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Freshwater Inflows in the Nueces Delta, TX: Impacts on Porewater Salinity and Estimation of Needs

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Freshwater Inflows in the Nueces Delta, TX: Impacts on Porewater Salinity and Estimation of Needs

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Abstract

Freshwater Inflows in a Texas Salt Marsh: Impacts on Porewater Salinity and Estimation of Needs

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Estuarine wetlands and salt marshes are fundamentally driven by variations in freshwater inflow. In semi-arid salt marshes, such as the Nueces River Delta, TX, the stochastic nature of freshwater inflow events exposes resident organisms to a wide range of environmental conditions. In this study, we investigate (1) the relative importance of environmental variables on porewater salinity and (2) determination of freshwater inflow needs based on the response of emergent plants to salinity variations. Porewater salinity variations were tracked on a continuous basis with deployed conductivity sensors and on a synoptic basis with soil water extracts. We found that spatial patterns of porewater salinity were characterized by a high degree of variability in creekbank areas (23.8 ± 7.68) relative to interior marsh areas (44.2 ± 3.4). Our observations were used to test a simple model capable of predicting porewater salinities based on environmental variables. Both empirical measurements and model simulations indicated that semiannual tides play a critical role in controlling porewater flushing from precipitation and freshwater inflow events.

Estimation of freshwater inflow needs for the Nueces Delta proceeded in two steps. First, we examined the response of three common emergent plants species (*Borrichia frutescens*, *Spartina alterniflora*, and *Salicornia virginica*) to variations in salinity. The abundance of one species in particular (*S. alterniflora*) was tightly coupled to salinity variations whereby salinities exceeding 25 ± 5 resulted in dramatic declines in coverage. Next, the relationship between freshwater inflow and porewater salinity was examined with respect to the salinity "tolerance" of *S. alterniflora*. Estimated inflow needs based on maintenance of substantial (> 20%) *S. alterniflora* coverage was comparable to both previous inflow needs estimates and mean annual inflows observed over the course of the study. The results of this study suggest that *S. alterniflora* abundance provides a reliable indicator of overall estuarine hydrological condition in the Nueces Delta.

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Chapter 1: Porewater Salinity Dynamics within the Creekbank Areas of an Irregularly Flooded Salt Marsh

Abstract

The zonation, physiology, and ecology of emergent salt marsh plants are largely controlled by variations in soil porewater salinity. The influence of porewater salinity on plants in regularly flooded salt marshes is well defined due to predictable diurnal or semidiurnal patterns of tidal inundation. However, the relative impact of environmental variables such as tidal period, precipitation, and freshwater inflow is not well understood in irregularly flooded salt marshes. Using a western Gulf of Mexico (Texas) salt marsh as an example, we examined the relative importance of various freshwater sources on porewater salinity. Conductivity sensors and energy-mass balance simulations provided evidence that porewater flushing was dependent on distance from tidal creeks as well as water level stage. Seasonal water level variations, corresponding with a semiannual tidal cycle, resulted in notable differences in inundation frequency between creekbank and interior marsh areas. Distinct patterns of plant zonation likely persist in this irregularly flooded salt marsh as a result of the periods of regular inundation that accompany the positive phase of the semiannual tidal cycle. We conclude that freshwater inflow events provide critical moderation of tidal creek salinities and flushing of sediment porewater. Severe droughts (like the one presently underway in south Texas) will result in further declines in freshwater inflow, reducing the frequency of low porewater salinity periods that are critical for maintenance of emergent plant communities. Loss of emergent

vegetation may ultimately result in reduced abundance of upper trophic level organisms that depend on this system for shelter, food, and habitat.

Introduction

Salt marsh organisms are exposed to multiple environmental factors including desiccation, UV radiation, and extreme temperatures. Among these, salinity is often cited as one of the most important determinants of primary production in low latitude estuarine systems [*Pennings et al.*, 2005; *Zedler*, 1983]. Generally, the negative effects of elevated salinity arise as a result of "physiological drought". Plants are unable to maintain water uptake from the soil due to lowered external water potentials [*Adam,* 1990]. Ultimately the degree to which organisms are exposed to these stressors depends on the frequency and duration of inundation by sea water [*Pennings and Callaway*, 1992; *Bertness and Ellison*, 1987]. Inundation frequency of a given site varies widely within and among salt marsh systems depending on its topography, exposure, and prevailing tidal pattern [*Silvestri et al.*, 2005]. Regularly flooded salt marshes are generally found in fringing estuaries with predictable diurnal or semidiurnal tides. By contrast, irregularly flooded salt marshes are generally found in areas where tidal creek water levels are driven by semiannual tides or meteorological forcing.

Regularly flooded salt marshes are characterized by a transition from fully marine conditions in low marsh areas to brackish or freshwater conditions in high marsh areas. This salinity transition is often associated with an elevation gradient [*Adam*, 1990; *de*

Leeuw et al., 1991]. Although some regularly flooded salt marshes in the Western Atlantic conform to this model remarkably well [*Bertness et al.*, 1992; *Pennings et al.*, 2005], other irregularly flooded salt marshes display little systematic variation in porewater salinity with respect to elevation [*Costa et al.*, 2003; *Noe and Zedler*, 2001; *Rasser*, 2009; *Silvestri et al.*, 2005]. In these irregularly flooded salt marshes, porewater salinity dynamics have been ascribed to variations in precipitation [*Dunton et al.*, 2001; *Noe and Zedler*, 2001], evapotranspiration [*Silvestri et al.*, 2005], and freshwater inflow [*Costa et al.*, 2003]. Few studies have examined the porewater salinity dynamics of irregularly flooded salt marshes [*Costa et al.*, 2003; *Silvestri et al.*, 2005] relative to regularly flooded salt marshes [*de Leeuw et al.*, 1991; *Morris*, 1995; *Moffett et al.*, 2010]. As a result, our ability to predict and model porewater salinity dynamics in irregularly flooded salt marshes is limited [*Pennings et al.*, 2005].

The porewater salinity dynamics of irregularly flooded salt marshes are difficult to characterize. Unpredictable inundation patterns make it difficult to design a sampling strategy capable of resolving significant but infrequent meteorological events [*Noe and Zedler*, 2001; *Pennings et al.*, 2003] and traditional sampling methods such as suction sampling and centrifugation are limited to areas of high soil moisture [*Rasser*, 2009]. In this paper we demonstrate the utility of a novel method for continuous measurement of porewater salinity in conjunction with traditional methods. Our observations were used to test a simple model capable of predicting porewater salinities based on environmental variables. We specifically test the hypothesis that freshwater inflow is the primary determinant of root zone porewater salinity in the Nueces River Delta, TX.

SITE DESCRIPTION

This study describes the porewater salinity dynamics of creekbank and interior marsh areas in the Nueces River Delta, TX (27º52´N, 97º32´W). The Nueces River Delta, located in South Texas, has a small tidal amplitude (~ 0.15 m) and generally low volumes of annual freshwater inflow and precipitation. The Nueces estuary, the second driest in Texas, has an average annual precipitation of only 76 cm y^{-1} [*Tolan*, 2007]. The delta is part of a negative estuary, where the hypersaline waters of Corpus Christi Bay mix with freshwater inflows from the Nueces River via the Rincon Bayou channel (Figure 1.1). Creekbank and interior marsh areas are subject to highly irregular flooding patterns driven more by meteorological conditions than lunar tidal cycles [*Bureau of Reclamation*, 2000]. This irregular flooding pattern occurs as a result of limited exchange with the Gulf of Mexico and filtering of the tidal signal (~15 cm amplitude) through the inlets to Corpus Christi Bay and the transition between Corpus Christi Bay and Nueces Bay [*Bureau of Reclamation*, 2000].

The delta covers an area of approximately 57 km^2 and is composed of a tidal creek network with vegetated creekbanks (Figure 1.1). Interior marsh areas are characterized by a mixture of vegetated and bare areas. The emergent plant community is dominated by perennial shrubs such as *Borrichia frutescens, Batis maritima,* and *Salicornia virginica* [*Forbes and Dunton*, 2006]. Previous research has shown that the local plant community depends on freshwater inflow and inundation to relieve stress and encourage seed germination [*Alexander and Dunton*, 2002]. However, freshwater inflows to the delta are

infrequent and limited to periods of Nueces River flow when water levels in the river are high enough to carry water through the Rincon Bayou channel which is elevated more than a meter above the Nueces River bed (*Bureau of Reclamation*, 2000; *Ward et al.*, 2002).

Although the emergent plant community has proved to be resilient to low salinity periods that occur as a result of flooding [*Alexander and Dunton*, 2002], the magnitude and frequency of freshwater inflow events has decreased substantially over the past 60 years as a result of municipal water management and local climate change [*Bureau of Reclamation*, 2000; *Forbes and Dunton*, 2006]. The recent decline in freshwater supply has led to dramatic reductions in the areal vegetative coverage of high marsh areas [*Forbes and Dunton,* 2006]. This study is motivated by the need to address management of water resources given the importance of freshwater inflow events to ecological function in the Delta [*Ward et al.*, 2002]. What is the volume of freshwater inflow needed to sustain ecological function? What is the potential for pipelines and river diversions to supplement ecological needs during drought and water deficit? These kinds of questions can only be addressed with confidence given a working knowledge of porewater salinity dynamics under a range of environmental conditions.

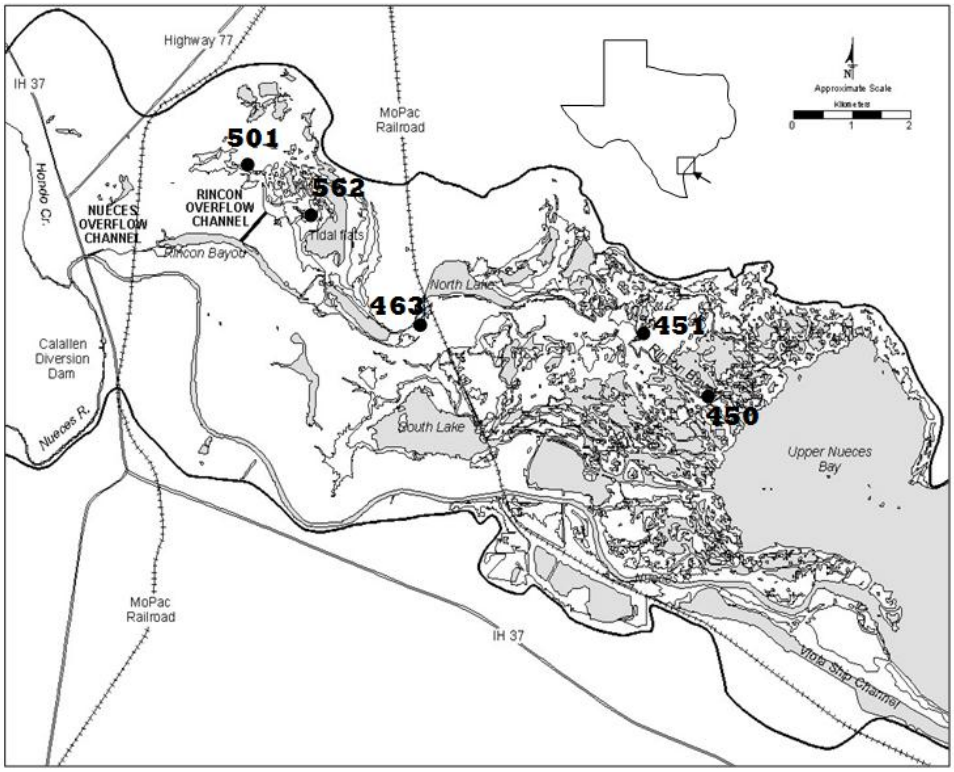

Figure 1.1: Location of sampling stations in the Nueces Delta.

Methods

POREWATER SALINITY MONITORING

A continuous record of porewater salinity was obtained using remotely deployed conductivity sensors (Model DST CTD, Star-Oddi Ltd., Reykjavik, Iceland). These sensors were buried at a depth of 20 cm below the sediment surface in creekbank and interior marsh areas at two sites (Site 450, Site 463, see Figure 1.1). A depth of 20 cm was chosen because this approximates the rooting depth of the two most common emergent vascular plants in the Nueces River Delta, *Borrichia frustescens* and *Salicornia virginica* [*Dunton et al.*, 2001]. The accuracy of the conductivity sensors was validated by comparison against porewater samples from sediment cores collected in the field. Porewater was extracted from these cores by centrifugation and measured with a handheld refractometer (Reichert Scientific Instruments, Buffalo, NY).

In order to supplement our continuous monitoring effort, we investigated the relative influence of various freshwater sources on long term records of porewater salinity (Table 1.1). Between 2001 and 2010 approximately 220 soil cores measuring 2.5 cm in diameter and 10 cm in length were collected quarterly from 5 sites in the Nueces River Delta (Figure 1.1). For this analysis, measurements in creekbank and interior marsh areas were pooled in order to evaluate the general importance of various freshwater sources.

Continuous porewater salinity records were compared against multiple environmental variables including precipitation, freshwater inflow, evaporation, tidal creek salinity, and tidal creek water level. Meteorological data including precipitation, wind speed, air temperature, and relative humidity was obtained from the Nueces Delta Weather Station located approximately 10 km from site 450 and 5 km from site 463. Tidal creek salinity information was obtained from stations NUDE2 and Salt08 for stations 463 and 450 respectively (available from the Division of Nearshore Research at Texas A&M – Corpus Christi, http://lighthouse.tamucc.edu/RinconSalinity). Both tidal creek salinity stations were located within 300 m of their respective study sites. Tidal creek water level data for site 463 was obtained from station NUDE2 while data for site 450 was obtained from the Nueces Bay tide gauge (available from the National Oceanic and Atmospheric Administration, http://tidesandcurrents.noaa.gov). A record of freshwater inflow to the Nueces River Delta was compiled from data collected by the Rincon Bayou Channel gauge (available from the U.S. Geological Survey, http://waterdata.usgs.gov/tx/nwis/dv?).

STATISTICAL ANALYSIS

We assessed the general importance of various freshwater sources on porewater salinity using a correlation matrix (Sigmaplot software, Version 10.0). For this analysis, porewater salinity and tidal creek salinity data from long term monitoring (2001 – 2010)

were reduced to quarterly means. Precipitation and freshwater inflow were aggregated into quarterly sums in order to facilitate comparison with salinity measurements.

The impact of precipitation events on porewater salinity was determined on the basis of inundation. The mean salinity change occurring as a result of precipitation events was evaluated for inundated and exposed sediments. Precipitation events were selected for analysis only if there was a continuous record of porewater salinity for 24 hours prior to the event. The effect of precipitation events on porewater salinity was calculated as the difference between the maximum salinity 24 hours prior to an event and the salinity minimum in following 24 hours. A 24 hour period was chosen because porewater salinities generally returned to their pre-event values by this time.

The influence of each group of precipitation events on porewater salinity (2010-2011) was investigated using a t-test (Systat v.13.0, Systat Software Inc., Richmond, CA, USA). The accuracy of model output with respect to empirically measured porewater salinity measurements was assessed using linear regressions (Sigmaplot software, Version 10.0). Significant differences were determined at $\alpha = 0.05$.

MODEL DESCRIPTION

A model describing the porewater salinity dynamics at selected sites in the Nueces Delta was constructed using the mass balance approach of *Tobias et al.* [2001] where the conservation of water and salt in wetland sediments is assumed dependent on water inputs by precipitation (Q_P), inundation (Q_T) and water losses due to

evapotranspiration (Q_{ET}) and creekbank drainage (Q_D) while the mass of salts in tidal creek and porewaters are represented by C_T and C_W respectively. The conservation of salt is described by the equation

$$\frac{(dS)}{(dt)} = Q_{G_W}C_{G_W} + Q_PC_P + Q_TC_T - Q_{E_T}C_{E_T} - Q_DC_W \quad (1)$$

where dS/dt is the rate of change in salt mass [*Tobias et al.* 2001]. Equation 1 was solved for porewater salinity (C_w) on a daily time step. Similar to *Tobias et al.* [2001], we assume that the concentration of salts in evaporate (C_{ET}), precipitation (C_P), and groundwater (C_{GW}) are negligible. Groundwater flux (Q_{GW}) was assumed to be negligible because we had no basis for estimating this value and our study area is not fringed by high elevation areas. While Q_P and C_T were measured empirically, the remaining model parameters were approximated using established methods. Daily evaporation (Q_{ET}, ET) was calculated using an approximation of the Penman-Monteith equation

$$\lambda ET = \frac{\left(\Delta(R_n - G) + \left(P_a C_p\left(\frac{(VPD)}{r_a}\right)\right)\right)}{\left(\Delta + \gamma\left(1 + \left(\frac{r_s}{r_a}\right)\right)\right)} \quad (2)$$

[*Allen et al.,* 1998] where the slope of the saturation vapor pressure curve (Δ) was obtained from a standard function relating saturation vapor pressure to air temperature. Mean air density (P_a) was calculated using the ideal gas law relating density to pressure

and temperature. Surface resistance (r_s) was taken as 70 s m^{-1} assuming that the Nueces Delta is adequately represented by the FAO standard reference crop where vegetation height is 0.12 m. This assumption is likely valid given the findings of *Heinsch et al.* [2004] that atmospheric exchange in Nueces Delta is typical of a dryland system. Aerodynamic resistance (r_a) was calculated as a function of measured wind speeds using the assumed standard reference crop. Vapor pressure deficit (*VPD*) was calculated as the difference between the saturation and actual vapor pressure approximated from relative humidity and air temperature measurements following *Allen et al.* [1998]. The psychometric constant value (γ) was set as 0.067 kPa°C^{-1} because measurements were taken at sea level.

Air temperature and relative humidity measurements were obtained from the Division of Nearshore Research at Texas A&M – Corpus Christi (available at http://lighthouse.tamucc.edu/RinconSalinity). Specific heat at constant pressure (C_p) and latent heat of vaporization (λ) were taken as constants (1.013E-3 MJ kg^{-1} °C^{-1} and 2.45 MJ kg^{-1} respectively). Soil heat flux (G) was taken to be zero [*Allen et al.*, 1998; *Wang et al.*, 2007]. Net radiation (R_n) was estimated using established relationships between latitude and extraterrestrial radiation [*Allen et al.*, 1998] and assuming maximum daily insolation [*Wang et al.*, 2007]. Established relationships between extraterrestrial radiation and latitude have a 1° resolution.

Modeling of sediment water balance (Q_T and Q_D) required the estimation or empirical determination of several additional parameters describing soil water drainage, sediment porosity, and inundation state (Table 2). Both infiltration and porewater

drainage depend primarily on hydraulic conductivity of the sediment matrix and the degree of saturation [*Dingman*, 2008]. Determination of porewater drainage for this study follows the method of Morris [1995] where specific yield (S_y) represents changes in porewater content as a function of sediment porosity and hydraulic head. Porewater drainage generally follows a predictable pattern whereby drainage increases with time until porewater volumes become depleted and soils approach their field capacity (F_c). Thus, F_c is a useful metric that can be used to insure that modeled salt and water fluxes are limited to realistic conditions. While sediment porosity was determined experimentally following *Percival and Lindsay* [1997], field capacity (F_c) was a model calibrated parameter. We conducted sensitivity analyses to test our assumptions regarding soil water drainage (specific yield, S_y) and field capacity (F_c) relative to published values from *Harvey et al.* [1987] and *Morris* [1995] respectively.

Inundation of creekbank areas in the Nueces Delta occurs irregularly. As a result, inundation state could not be reliably estimated based on lunar tidal cycles. This study did not benefit from the use of piezometers. As an alternative, inundation of creekbank areas by tidal creek waters was estimated by repeated visual observation over the course of the study period. Using our observations and nearby water level gauge measurements, we identified a water level stage at which the creekbank sediments at each site became inundated. The model takes these water level stage values as input to determine sediment inundation state (Table 2).

Extended periods of emersion produced situations where salts were present in the soil matrix in precipitated rather than dissolved form. Salt precipitation and dissolution

was simulated using the *Manganaro and Schwartz* [1985] and *Lionberger et al.* [2004] energy-mass balance models.

$$R = kA_d(C_1 - C_2)M \qquad (3)$$

The rate of crystallization or dissolution was determined as the product of the molecular weight of salt (M), the wetted area (A_d), the mass transfer coefficient for salt dissolution (k) and the difference between the concentrations of salt in solution (C_1) minus the saturated concentration at a given temperature (C_2). An important assumption of these models is that diurnal variation in environmental variables is unimportant. Therefore, only diurnal average values are taken as model input.

Results

Field Measurements

Long term porewater salinity measurements obtained via sediment cores and pooled from both creekbank and interior marsh areas were strongly correlated ($r^2 = 0.83$, $p < 0.05$) with tidal creek salinity. Although precipitation did not appear to be directly linked to long term porewater salinity measurements (Table 1.1), continuous monitoring revealed that their influence is highly dependent on water level (Figure 1.3). This may indicate that coarse temporal sampling (monthly - quarterly) is not sensitive enough to

detect the daily influence of precipitation events. Water level measurements showed distinct seasonal differences (Figure 1.2C). While high water levels generally occurred during spring and fall, low water levels occurred in winter and summer.

Continuous porewater salinity monitoring revealed that creekbank salinities are highly variable compared to interior marsh areas (Figure 1.2A, 1.2B). An extended data gap in porewater salinity measurements occurred from November 2010 to March 2011. Prior to this gap, mean porewater salinity in creekbank areas was 23.84 ± 7.68 while porewater salinity in interior marsh areas was 44.20 ± 3.42. Missing data is attributed to a lack of tidal flooding and near complete drying of the root zone sediments. Gravimetric soil water content measurements (n =12) taken during this period revealed that soils were extremely dry (< 55%). Subsequent laboratory tests (n=3) revealed that sensors deployed in soils of less than 42% gravimetric soil water content failed to recover porewater salinity data.

Two large freshwater inflow events occurred over the course of this study, allowing us to separate the effects of precipitation and freshwater inflow on tidal creek salinity. Precipitation events appeared to affect the porewater salinity of both creekbank and interior marsh areas whereas the impact of freshwater inflow events was limited to creekbank areas (Figure 1.2A-B, 1.2D-E). Porewater salinities remained unchanged in 2011 when there was a lack of either precipitation or freshwater inflow events. This suggests that local precipitation causes only minor short term variability (hours to days) in porewater salinity while freshwater inflow events cause critical flushing of sediments that slows or reverses the development of hypersaline marsh areas. Our hypothesis is

supported by the dependence of porewater salinity dynamics on tidal creek salinities which are dramatically impacted by freshwater inflow events (Table 1.1).

Table 1.1: Simple correlation (r) relationships between freshwater sources and salinity parameters. All salinity terms represent quarterly measurements taken between 2001 and 2010 in both creekbank and interior marsh areas. Significant interactions ($\alpha = 0.05$) are denoted by an asterisk.

	Precipitation	Freshwater Inflow	Creekbank Porewater Salinity
Precipitation (mm)	-		
Freshwater Inflow ($m^3\ day^{-1}$)	-0.08	-	
Porewater Salinity (‰)	-0.36	-0.14	-
Tidal creek Salinity (‰)	-0.42*	-0.16	0.91*

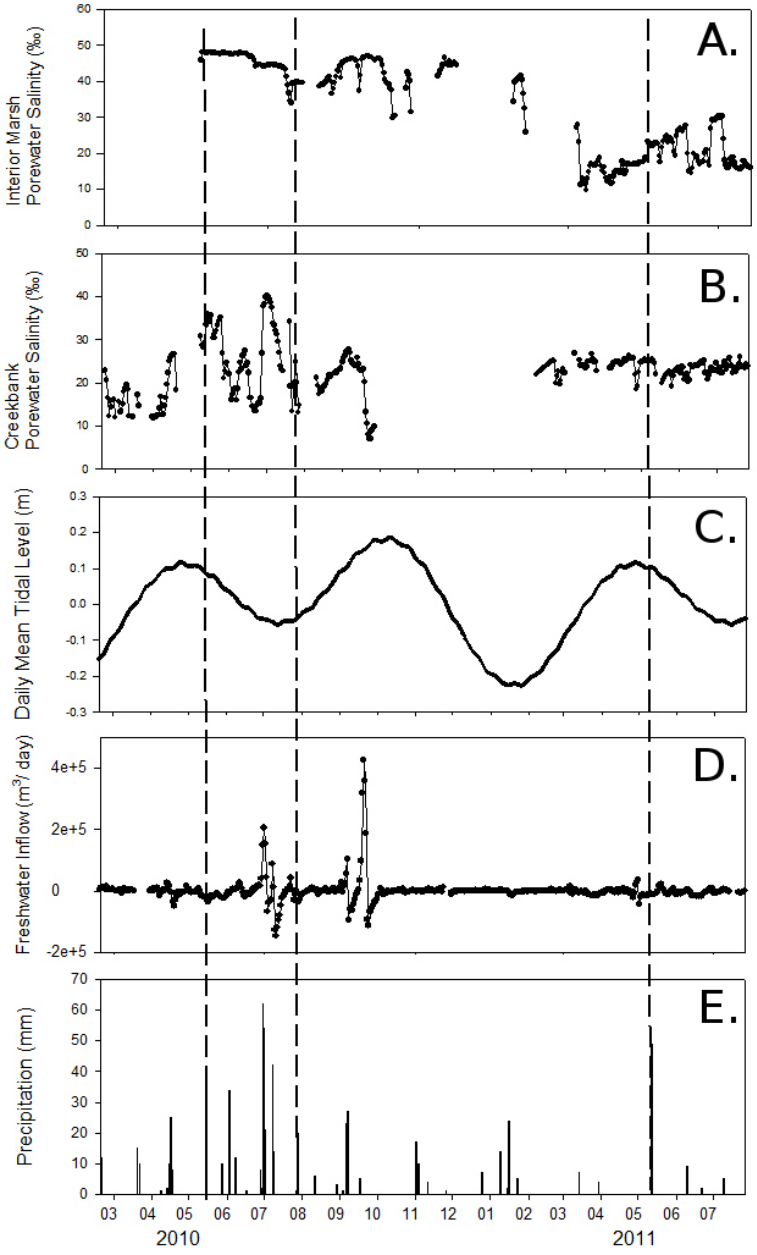

Figure 1.2: Times series of porewater salinity at site 450 (A and B), mean water level in Nueces Bay (C), freshwater inflow to the Rincon Bayou (D), and precipitation (E). Local precipitation data was recorded at the Nueces Delta weather station (NUDEWX). Freshwater inflow (discharge) data was taken at the USGS Rincon Bayou gage station (#08211503). Gaps in the porewater salinity data occurred as a result of low soil moisture conditions. Dashed lines highlight precipitation events not accompanied by a freshwater inflow event.

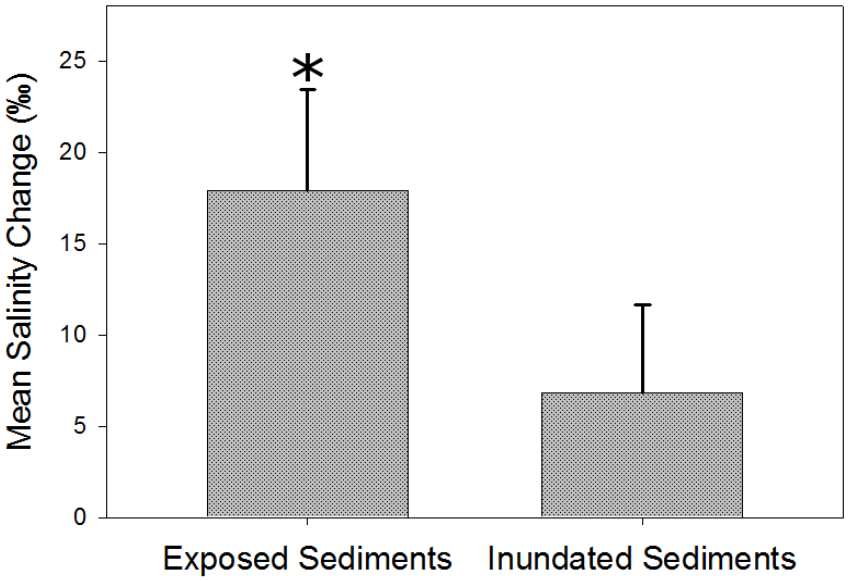

Figure 1.3: The effect of precipitation events on the porewater salinity of interior marsh areas calculated as a function of tidal creek water level. Significant ($\alpha = 0.05$) precipitation effects are denoted by an asterisk.

MODEL RESULTS

Model simulations were limited to creekbank areas because of high porewater salinity variability at the creekbank relative to interior marsh areas. We had greater confidence in our ability to estimate surface water – porewater interactions in creekbank areas where inundation occurs more frequently. The mass balance model output was compared to porewater salinity monitoring data (Figure 1.4). The model was found to have good agreement with field measurements especially during periods when the marsh was inundated by seasonal high tides ($r^2 = 0.59$, $p < 0.05$) while poor agreement was found during periods of extended emersion ($r^2 = 0.09$, Figure 1.4). The model accurately captured the timing and magnitude of salt crystallization and dilution events, which was evident from porewater salinity values increasing following the rewetting of soils (Figure 1.4). Our choice of inundation stage (Table 1.2) for each site resulted in a flooding frequency for sites 450 and 463 of 42.2% and 17.8% respectively. Model sensitivity analyses showed little effect of changing sediment field capacity (F_c) but a large dependence on specific yield (S_y) (Figure 1.5).

Table 1.2: Model constants describing physical and hydraulic properties of sediments.

Constant	Value	Source
Specific Yield	0.12	Tobias et al. (2001)
Porosity	0.67	This Study, experimentally determined
Site 450 Inundation Stage (m)	0.05	Personal Observation (NOAA Tide Station #8775244)
Site 463 Inundation Stage (m)	0.54	Personal Observation (DNR, Station 042)
Field Capacity	0.90	This Study, model calibrated

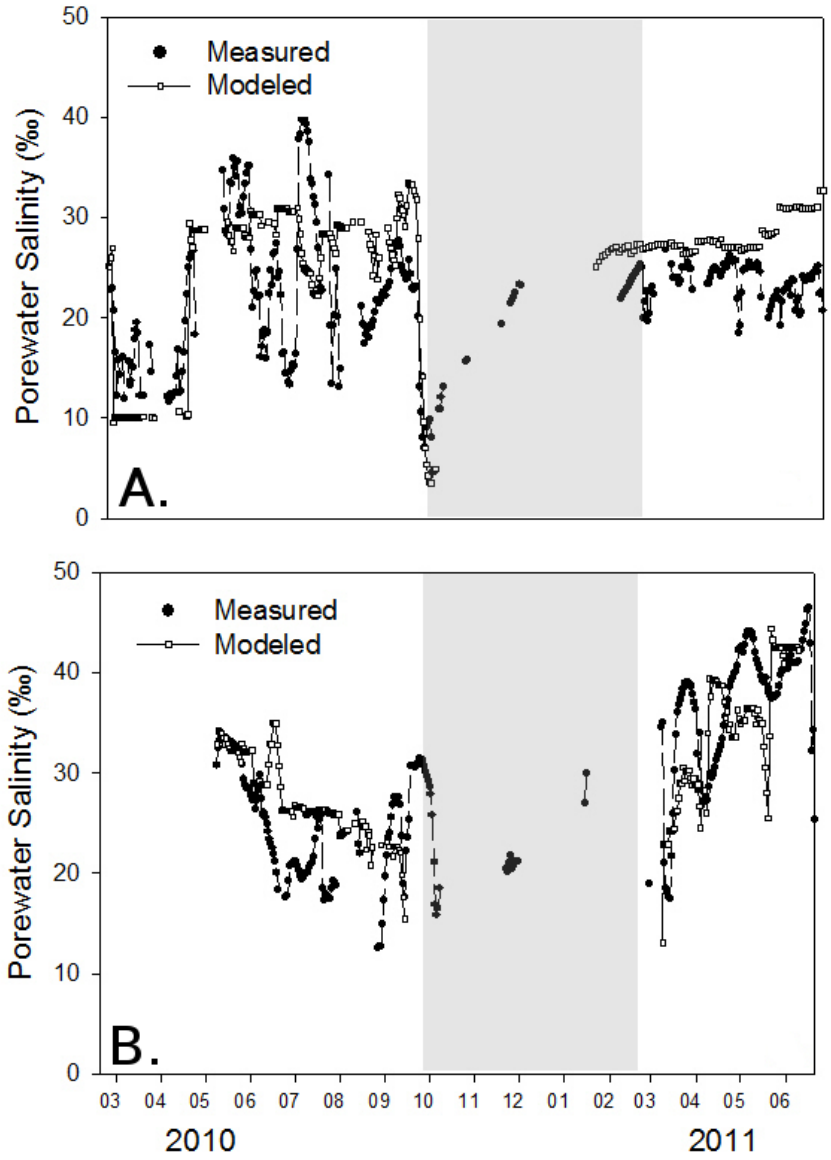

Figure 1.4: Comparison between modeled and measured porewater salinities at (A) site 450 (r^2= 0.41) and (B) site 463 (r^2= 0.48). Shaded portion of graphs indicate periods when measured data was missing due to extremely dry sediments.

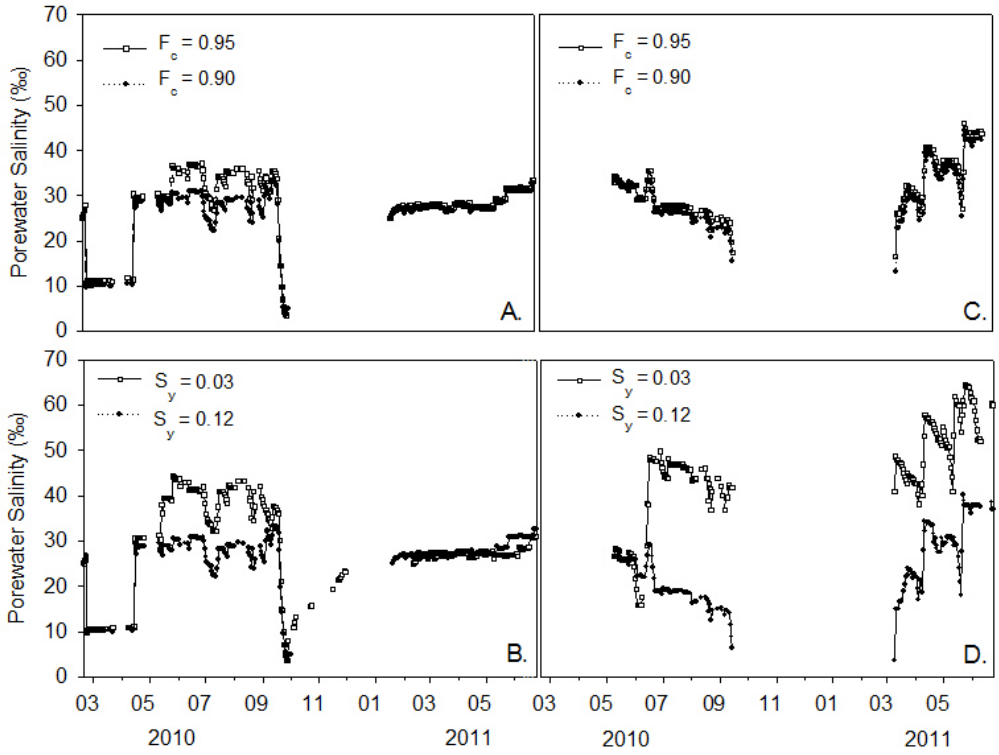

Figure 1.5: Model sensitivity analysis for station 450 (A, B) and station 463 (C, D) with respect to (A, C) sediment field capacity (Fc) and (B, D) soil drainage rate (specific yield, Sy). Values for this study (Sy = 0.12, Fc = 0.90) describe soils with high drainage rates and low field capacities relative to salt marsh soils in Virginia [*Harvey et al.* 1987] and South Carolina [*Morris* 1995].

Discussion

SPATIAL DISTRIBUTION OF VEGETATION IN RELATION TO POREWATER SALINITY

One of the difficulties in assessing the physiological tolerance of emergent salt marsh plants relative to short term environmental variations is that established monitoring methods are insensitive to the diurnal variation of driving variables such as precipitation, tidal inundation, and evapotranspiration [*Noe and Zedler*, 2001]. Characterization of porewater salinity dynamics within salt marshes usually involves the collection of porewater by suction sampling of monitoring wells [*Costa et al.,* 2003; *Morris*, 1995] or from sediments collected in the field [*Alexander and Dunton*, 2002]. Studies that involve the collection of soil cores estimate porewater salinity either by freshwater dilution and measurement of the supernatant [*Pennings et al.*, 2003] or through porewater extraction by centrifugation [*Alexander and Dunton*, 2002]. However, extrapolation of porewater characteristics measured on short temporal scales at selected sites make it difficult to interpret variability in marsh plant species composition and abundance on a system-wide basis [*Noe and Zedler*, 2001].

The development of remote sensing applications such as electromagnetic induction [*Moffett*, 2010], have enabled ecologists to at least partially address this challenge. This technique effectively characterizes spatial patterns of porewater salinity, but only provides a temporal snapshot of environmental conditions [*Rhoades et al.*, 1999]. Continuous monitoring of salinity using remotely deployed conductivity sensors

as presented here provides a potential solution to problems of sampling frequency. However, one limitation of continuous monitoring is that extremely high porewater salinities are often not captured due to the limited ability of sensors to recover conductivity information at low soil moistures (Figure 1.2). In this study, sensors were unable to provide reliable measurements of porewater salinity when gravimetric soil moisture content decreased below 55%. However, porewater salinity measurements taken throughout the study period from centrifuged soil cores were in the same range (10 – 50 ‰) as continuous measurements.

Continuous measurements of porewater salinity may help resolve longstanding questions in salt marsh ecology regarding the relative importance of environmental factors to the population dynamics of emergent plants. For example, studies examining the zonation of emergent salt marsh plants have revealed that zonation patterns often correspond with spatial patterns in sediment saturation and aeration [*Costa et al.*, 2003; *Pennings et al.*, 1992; *Pennings et al.*, 2005; *Silvestri et al.*, 2005]. However, the influence of salinity, oxygen availability, nutrient availability, and soil moisture to the population dynamics of emergent plants is currently unresolved. This had led some researchers to conclude that population dynamics are controlled by a combination of factors [*Silvestri et al.*, 2005].

Previous investigations have noted conspicuous zonation of emergent salt marsh plants according to environmental characteristics that vary according to elevation gradients [*Rasser*, 2009; *Wang et al.*, 2007]. However, in the Nueces Delta, lack of significant differences in porewater salinity, soil moisture, or porewater nutrients among

vegetation zones led *Rasser* [2009] to conclude that environmental factors may fluctuate on smaller spatial or temporal scales than measured in his study [e.g. see *Noe and Zedler*, 2001]. We present evidence for porewater salinity variations on much shorter temporal scales (hours to days). Our observations of high porewater salinity variability in creekbank areas relative to interior marsh areas is consistent with the hypothesis of *Harvey et al.* [1987] who predicted that the porewaters of creekbank areas experience rapid turnover compared to interior marsh areas.

Differing porewater salinities in creekbank and interior marsh areas may control emergent plant distributions in the Nueces Delta [*Forbes and Dunton*, 2006; *Rasser* 2009]. For example, *Spartina alterniflora,* which is only found at extremely low elevations adjacent to creekbanks, has a low tolerance for extreme variations in porewater salinity [*Touchette et al*., 2009]. A study by *Webb* [1983] on Galveston Island, Texas found that porewater salinities exceeding 25‰ resulted in significant reduction in density, height, and standing biomass of *S. alterniflora.* The *S. alterniflora* root zone is generally buffered from extreme variation due to consistent inundation with tidal creek water. The root zone of *Borrichia frutescens,* in contrast, is found on elevated creekbank levees where sediments are irregularly inundated and porewater salinity is highly variable (Figure 1.2). Porewater salinity dynamics in interior marsh areas, which are dominated by *Batis maritima,* are generally stable except for small short term variations (Figure 1.2). Future studies should assess whether these species have differing physiological responses to variations in porewater salinity occurring at a range of temporal scales.

Another challenge in determining the ecological importance of porewater salinity fluctuations is the difficulty in accounting for fine scale vertical structure [*Casey and Lasaga,* 1987]. This study examined porewater salinity dynamics at a relatively coarse vertical scale (20 cm). One potential disadvantage of this approach is that fine scale vertical patterns in porewater salinity are not resolved. However, our results remain useful as a first approximation of the plant community's exposure to fluctuations in porewater salinity. Unfortunately, the extent to which differences in rooting depth control plant species' distributions in the Nueces Delta is unknown [*Rasser*, 2009]. It is likely that species with a shallow root zone such as *B. maritima* respond more strongly to small precipitation events where porewater flushing is limited to the top several centimeters of sediment while species with a deep root zone may only benefit from the large magnitude flushing events that accompany tidal creek inundation. Alternatively, species with a deep root zone may be adapted for survival at high elevations such as atop creekbank levees where deep penetrating roots are necessary to tap deeper groundwater [*Bonin and Zedler*, 2008]. Under this scenario, increases in drought frequency would likely lead to a decrease in the abundance of deep rooted species in favor of shallow rooted stress-tolerant species [*Forbes and Dunton,* 2006]. Future studies, resolving this fine scale structure, have the potential to predict the trajectory of future plant community assemblages as functions of climate, rooting depth, and physiological tolerance to fluctuations in porewater salinity.

SALINITY AS AN INDICATOR OF REPLACEMENT AND DILUTION OF SEDIMENT POREWATER

Although this study focuses on porewater salinity because of its demonstrated importance in controlling the abundance and distribution of plants in low latitude salt marshes, variations in porewater salinity are broadly reflective of porewater discharge and replacement. Past studies in the Nueces Delta have noted a tight coupling between freshwater inflow events and tidal creek salinities [*Alexander and Dunton*, 2002; *Forbes and Dunton*, 2006; *Ward et al.*, 2002]. Our results are consistent with these earlier studies regarding the relationship between tidal creek and porewater salinities. However, we found that under seasonal low tide circumstances precipitation is often the dominant source of porewater flushing (Figure 1.3). We were also able to assess the relative importance of various freshwater sources on porewater replacement and porewater salinity. Our results show that the influence of environmental factors on the porewater salinity dynamics of the Nueces Delta, is highly dependent on water level variations (Figure 1.2, Figure 1.3). It appeared that inundation patterns and the resulting variations in porewater salinity were highly correlated with a 6 month frequency known as the secular semiannual "tide" [*Ward et al.*, 2002]. High water levels, present in fall and spring, resulted in more frequent inundation and porewater flushing.

We used the results of our modeling exercise to evaluate potential replacement and dilution of porewaters. For example, sensitivity analyses indicate that creekbank drainage in the Nueces marsh occurs at a high rate compared to salt marsh soils elsewhere (Figure 1.5). In this study, porewater replacement was assumed to take place

primarily through vertical percolation. This assumption is likely valid because other modeling studies in Virginia, Italy and South Carolina, which directly measured water fluxes, have found that lateral diffusion of water through tidal creekbanks occurs at a much slower rate than vertical infiltration [*Harvey et al.*, 1987; *Ursino et al.*, 2004; *Gardner*, 2005]. Our inundation stage estimates (Table 1.2) allowed us to assess the accuracy of our assumption through its incorporation into our mass balance model. For example, porewater salinity fluctuations following high water levels exceeding inundation stage were apparent in both modeling output and *in situ* measurements (Figure 1.2, 1.3). In addition, our estimates of inundation frequency (20-40%) were consistent with studies in other irregularly flooded salt marshes [*Costa et al.*, 2003]. Our results provide further support to the idea that freshwater inflow events per se are not important but rather inundation (overbanking) events [*Ward*, 1985].

Model output was consistent with empirical porewater salinity measurements despite numerous assumptions regarding evapotranspiration, groundwater flux, and soil hydraulic conductivity. In addition, model results were realistic with respect to the relative importance of various model inputs [*Morris*, 1995]. For example, inundation frequency was a major determinant of porewater salinity. In contrast to both our expectations and previous research, evapotranspiration was not a dominant factor driving porewater salinity dynamics [*Pennings et al.*, 2005]. Future research should focus on obtaining accurate estimates of factors controlling surface water interactions rather than those controlling atmospheric water interactions. Modeling of porewater salinity could be improved with more detailed estimates of factors describing the interaction between tidal

creek water and sediment porewater such as inundation frequency and soil hydraulic conductivity. This is likely to be especially important in systems with creekbanks of variable elevation composed of sediments with diverse hydraulic properties [*Morris*, 1995].

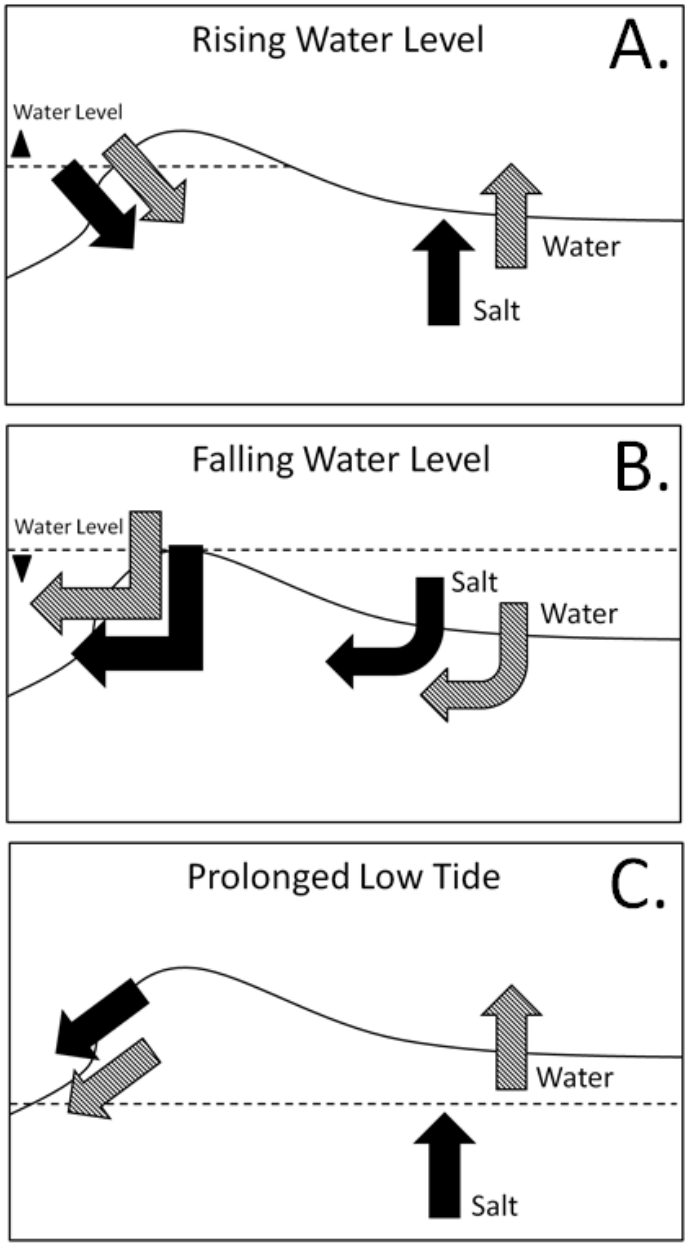

Figure 1.6: Conceptual diagram depicting salt (black arrows) and water (shaded arrows) fluxes within creekbank and interior marsh areas. Average salt and water transport is depicted for situations where (A) tidal creek water levels are rising, (B) falling, or (C) sustained at a low level during seasonal low tides. Areas below tidal creek water level (dashed lines) are assumed to be inundated.

The relative importance of freshwater inflow events on porewater salinity dynamics in creekbank versus interior marsh areas and its dependence on tidal creek water level is depicted in a conceptual model (Figure 1.6). This conceptual model is a time series showing infiltration and discharge over a complete range of tidal creek water level stages beginning with rising water levels below inundation stage (Fig. 5A), progressing towards inundation (Fig. 5B) and ending with discharge at low water level (Fig. 5C). First, tidal creek stage rises within adjacent creeks and creekbank areas receive salt and water inputs (Fig 5A). Second, sediments are inundated by tidal creek flood waters (Fig. 5B) causing salt and water to enter creekbank and interior marsh soils via vertical percolation. During this period, porewater replacement occurs more rapidly in creekbank areas than in interior marsh areas. Third, seasonal low tides cause extensive draining of creekbank soils (Fig. 5C). As long as water levels remain below inundation stage and soil moistures remain high (exceeding field capacity), creekbank and interior marsh soils lose water due to evaporation (Fig. 5A, C). As a result, salts concentrate in exposed surface sediments [Fig. 5A, 5C; *Casey and Lasaga*, 1987].

Our conceptual model is consistent with some aspects of porewater replacement observed in other regularly inundated salt marshes in Virginia [*Harvey et al.*, 1987] and South Carolina [*Morris*, 1995] whereby fluctuating tides control discharge and recharge of sediment porewater. However, in the irregularly flooded areas of the Nueces marsh prolonged low water levels in summer and winter lead to extensive evaporation and drainage of creekbank and interior marsh areas. Porewater replacements in both the Nueces marsh and irregularly flooded salt marshes elsewhere are driven by unpredictable

daily and seasonal variations in estuary water level [*Costa et al.*, 2003]. However, in contrast to these other areas, distinct zonation of emergent plants persists in the Nueces marsh despite extreme variations in porewater salinity [*Rasser*, 2009].

We believe that alternating periods of regular and irregular inundation produced in response to the phases of the semiannual secular tide is the primary determinant of zonation patterns in the Nueces Delta. During the negative phase of the semiannual tide, the marsh surface is often exposed for long periods of time (weeks to months). Both low water levels during these periods and extreme flooding events discourage processes that produce distinct zonation patterns. In contrast, periods of regular inundation produce differences between the environmental characteristics of creekbank and interior marsh areas and encourage distinct patterns of plant zonation. This hypothesis is supported by *Forbes and Dunton* [2006] who noted that the greatest emergent plant diversity in the Nueces marsh was found during moderate climactic periods. During extremely wet periods, the plant community was dominated by only a few ecologically dominant species. Conversely, during dry periods the plant community was reduced to only a few species capable of tolerating extreme environmental stress. Thus, porewater salinity dynamics in the Nueces marsh share some characteristics with salt marshes elsewhere but are uniquely controlled by the conspicuous semiannual secular "tide" [*Ward et al.*, 2002].

The relative effects of various freshwater sources on the porewater salinity of salt marsh sediments is likely to vary among estuarine systems according to local and regional hydrography/meteorology. We conclude that, in the Nueces Delta, freshwater inflow events provide critical moderation and flushing of sediment porewaters. The

importance of these events is especially apparent during drought years when the absence of freshwater inflow leads to hypersalinity and extreme soil moisture deficit [*Forbes and Dunton,* 2006]. A substantial portion of this study encompassed a severe drought period beginning in May, 2011 that intensified following the conclusion of monitoring in July, 2011. Palmer Drought Severity Index values at the conclusion of monitoring (< -2.75) indicated that South Texas was in an exceptional drought (see National Climate Data Center, http://www.ncdc.noaa.gov/sotc/drought/2011/7).

Further declines in freshwater inflow due to a combination of municipal water withdrawals and global climate change are likely to reduce the frequency of low porewater salinity periods that are critical for maintenance of emergent plant communities. Similar studies incorporating continuous monitoring of porewater salinity are needed in order to estimate the potential negative impact of municipal water withdrawals from upstream water sources as well as the potential for river diversions and other mitigation efforts to offset negative anthropogenic impacts.

Chapter 2: Estimation of Freshwater Inflow Requirements for a Semi-arid Salt Marsh Using Emergent Plants as Indicators of Ecosystem Condition

Abstract

Estuarine wetlands and salt marshes are fundamentally driven by variations in freshwater inflow. In semi-arid salt marshes, the stochastic nature of freshwater inflow events exposes resident organisms to a wide range of environmental conditions. Estuaries with heavily modified hydrology due to upstream dams and municipal water use are particularly vulnerable to climatic extremes. Assessment of anthropogenic impacts on the health of estuarine wetlands has traditionally focused on economically important shellfish and finfish species. In this study, we develop a novel method for determination of freshwater inflow needs based on emergent plants as indicators of ecosystem condition. The impact of freshwater inflow events on three common emergent plants in the Nueces River Delta (*Spartina alterniflora*, *Borrichia frutescens*, *Salicornia virginica*) was determined from long term monitoring of permanent census plots. Of the three species examined, *Spartina alterniflora* was determined to be the best indicator species because its abundance most closely tracked variations in freshwater inflow. For example, under low salinity conditions *S. alterniflora* cover approached 66%. However, when salinities exceeded 25, *S. alterniflora* cover declined rapidly. Estimates of freshwater inflow needs using *S. alterniflora* were comparable with estimates obtained in previous studies using more complex and more expensive methodologies such as fisheries based optimization modeling. Our results provide clear evidence that the presence or absence of key plant indicator species (in this case *S. alterniflora*) is reflective of overall estuarine hydrological condition over time scales exceeding six months.

Introduction

Estuarine wetlands and salt marshes are fundamentally driven by variations in freshwater inflow. The frequency, duration, and seasonal distribution of these "freshets" often determine the physiochemical characteristics of both aquatic and sedimentary wetland environments [*Zedler* 1983; *Mitsch and Gosselink* 2007]. Environmental characteristics such as nutrient concentration and salinity are regulated by freshwater inflow events and ultimately restrict the distribution and abundance of estuarine organisms [*Adams* 1963; *Alexander and Dunton* 2002; *Mitsch and Gosselink* 2007]. The relative impact of freshets within a particular estuary is dependent on the tidal regime, precipitation frequency, geomorphology, and water residence time [*Solis and Powell* 1999; *Brock* 2001]. Small freshwater inflow events are capable of flushing estuaries with small water volumes and large tidal ranges. However, the waters within microtidal semi-arid estuaries exhibit long residence times and require large freshwater inflow events to effectively flush accumulated salts and nutrients from sediments [*Solis and Powell* 1999].

The Nueces Estuary, near Corpus Christi, TX, represents one of the largest, driest, and least flushed estuaries along the Gulf of Mexico [*Solis and Powell* 1999]. Although large freshwater inflow events are relatively rare in this system, their occurrence significantly impacts physiochemical characteristics and biological communities [*BOR* 2000; *Alexander and Dunton* 2002; *Powell et al.* 2002; *Montagna et al.* 2009]. Freshwater inflow events in the Nueces Estuary are also subject to a high degree of interannual and interdecadal variability. Increased freshwater inflow during wet years

increases the abundance and physiological condition of emergent vegetation, ichthyoplankton, and benthic infauna [*Montagna et al.* 2002; *Forbes and Dunton* 2006; *Tolan* 2008]. In contrast, salinity stress and moisture deficits common during dry years, results in the decreased abundance and altered community structure of resident estuarine organisms [*Forbes and Dunton* 2006; *Montagna et al.* 2009].

The construction of upstream reservoirs, intended to increase municipal water supplies, has resulted in significant alterations to the Nueces Estuary [*BOR* 2000]. Several environmental impact assessments followed the construction of the Lake Corpus Christi (1958) and Choke Canyon (1982) reservoirs [*BOR* 1975; *TDWR* 1982; *Pulich et al.* 2002]. These studies were intended to document the impact of reservoir development on downstream ecosystems and estimate future freshwater inflow needs. Estimating freshwater needs in the Nueces Estuary is confounded by extreme interannual variations in freshwater supply coupled with dramatic hydrologic changes to the watershed resulting from reservoir construction. For example, freshwater inflows to the Nueces Delta have decreased by approximately 99% in combination with a decline in the average magnitude of flood events relative to pre-reservoir conditions [*Ward and Irlbeck* 2000].

Freshwater inflow requirements are often determined by the physiological requirements of several "focal" or "indicator" species [*TDWR* 1982; *Longley* 1994; *Doering et al.* 2002; *Pulich et al.* 2002; *Richter et al.* 2003; *BBEST* 2011]. These indicator species are selected because they are either economically important or particularly sensitive to environmental conditions [*Dale and Beyeler* 2001; *Doering et al.* 2002]. Following the identification of an indicator species, field observations are used to

determine its critical salinity threshold. After the salinity tolerances of a suite of indicator species have been determined, they are related to freshwater supply, and used to estimate specific inflow requirements.

It is important to note that, in this study, the term "indicator species" refers to a *condition* indicator rather than a *composition* indicator. While a composition indicator is used as a proxy for a distinct species assemblage, condition indicators are used as a proxy for a distinct set of environmental conditions [*Zacharias and Roff* 2001]. Condition indicators are selected for their ability to track fluctuations in environmental conditions and can be used to monitor changes in habitat quality as a result of management practices [*Zacharias and Roff* 2001]. This is consistent with many conservation programs, which seek to limit their focus to maintaining representative habitats rather than maximizing specific productivity or biodiversity metrics [*Palmer et al.* 1997; *Mitsch and Gosselink* 2007]. Although this approach is widely used, it may fail to take into account the impact of functional redundancy among species and it assumes that species responses are independent from one another [*Ter Braak and Prentice* 1988; *Palmer et al.* 1997]. These limitations are partially addressed through the selection of a suite of indicator species. The use of multiple condition indicators is assumed to account for unknown environmental variables as well as potential dependency among species [*Ter Braak and Prentice* 1988].

Invertebrates, such as the blue crab (*Callinectes sapidus*) and commercially important fish species such as the Atlantic croaker (*Micropogonias undulatus*), are often used as indicators of estuarine ecosystem condition [*Powell et al.* 2002; *BBEST* 2011].

The use of these species as indicators is only possible because of intensive monitoring programs (e.g. Texas Parks and Wildlife Fishery-Independent Monitoring Program, *Pulich et al.* 2002, *Buzan et al.* 2009). However, it is currently unclear whether these species provide a reliable representation of environmental conditions because they experience high population variability, incur losses due to fishing pressure, and are subject to seasonal migration [*Dale and Beyeler* 2001; *Powell et al.* 2002].

In contrast to nekton species, vascular marsh plants are immobile and are not normally subject to harvesting pressures. In estuaries, plant zonation and distribution is largely controlled by soil porewater conditions rather than tidal creek water [*Bertness et al.* 1992]. Because porewaters have longer residence times, rooted plants reflect environmental conditions over longer time scales. Several studies have examined the utility of vascular plants as estuarine indicators. For example, submerged vascular plants have been used as condition indicators to estimate freshwater inflow needs in Florida [*Doering et al.* 2002]. Although emergent plants are infrequently used to estimate freshwater inflow needs within Texas estuaries, they satisfy established criteria for use as indicator species [*Dale and Beyeler* 2001] and have been developed as indicators of ecosystem condition in Georgia [*White and Alber* 2009].

The objective of this study was to evaluate the utility of emergent plants as indicators of ecosystem condition and freshwater inflow requirements for the Nueces River Delta, TX. The response of the overall plant community to variations in freshwater inflow was used to determine whether the plant community exhibited a consistent response to hydroclimatic periods. Next, the response of individual plant species to

freshwater inflow events was addressed by 1) determining the salinity tolerance of potential indicator species and 2) deriving the relationship between freshwater inflow and porewater salinity. This study specifically investigated the hypothesis that smooth cordgrass (*Spartina alterniflora*) abundance reflects variations in freshwater inflow and subsequent variations in porewater salinity. Our salinity tolerance determinations for emergent plants improves on earlier studies which were generally limited to time periods of less than three years and in some cases were established from only a single survey [*Penfound and Hathaway* 1938; *Adams* 1963; *Webb* 1983].

Methods

This study was conducted in the Nueces River Delta (27° 51’ N, 97° 31’ W) located in the Northwestern Gulf of Mexico. The Delta is comprised of an expansive complex of tidal flats bisected by a tidal creek network (Figure 2.1). The low marsh plant assemblage is dominated by ox-eye daisy (*Borrichia frutescens*), glasswort (*Salicornia virginica*), and saltwort (*Batis maritima*). Tidal creeks are fringed with stands of smooth cordgrass (*Spartina alterniflora*) and high marsh areas are dominated by expansive gulf cordgrass (*Spartina spartinae*) meadows [*Rasser* 2009].

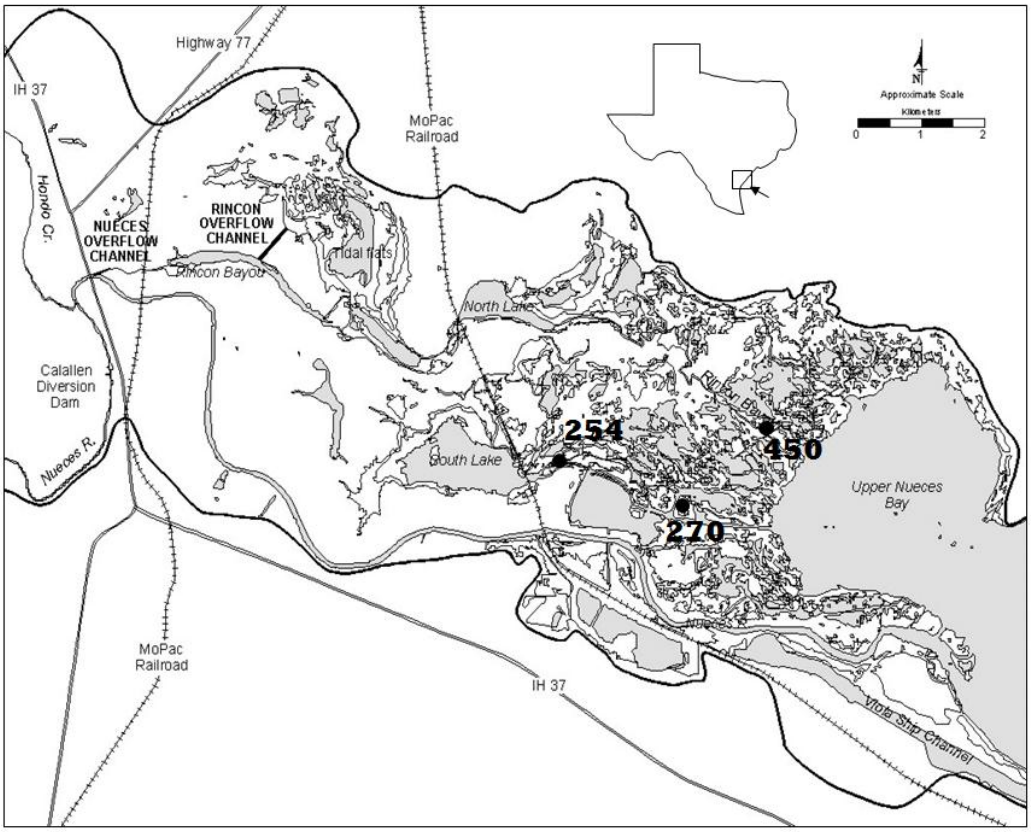

Figure 2.1: Location of sampling stations in the Nueces Delta.

HYDROGRAPHY

The Nueces Delta is located within a semi-arid region of low average annual precipitation (76 cm y^{-1}). Dry conditions persist throughout most of the year except following rare tropical storm events that develop in late summer. The hydroclimatic regime has a marked seasonal pattern due to the pulsed nature of freshwater inflows (Figure 2.2) which inundate the Delta when flows in the Nueces River exceed approximately 2.16 x 10^6 m^3d^{-1} [*Montagna et al.* 2009]. When flows fail to exceed this threshold, Nueces River discharge moderates wetland salinities indirectly by lowering the salinity of Nueces Bay. Freshwater inflows to the Nueces Estuary over the past 20 years (1990-2011) exhibit highly variable patterns (Figure 2.2). During the study, the Delta experienced both extremely dry conditions during the period from 2008 to 2011 and extremely wet conditions during the period from 2002 to 2005. Freshwater inflows were high during the study period relative to the past 20 years, but relatively low compared to historic levels (Figure 2.3). Wet and dry periods were identified using a simple statistical analysis of Nueces River flows throughout the study period (1999-2011). Drought periods were defined as years with inflows below the median (Figure 2.3, Table 2.2). Although direct precipitation can potentially affect environmental conditions in the Delta [*Dunton et al.* 2001], it was not included in the analysis of drought periods because there was no consistent relationship between precipitation and salinity.

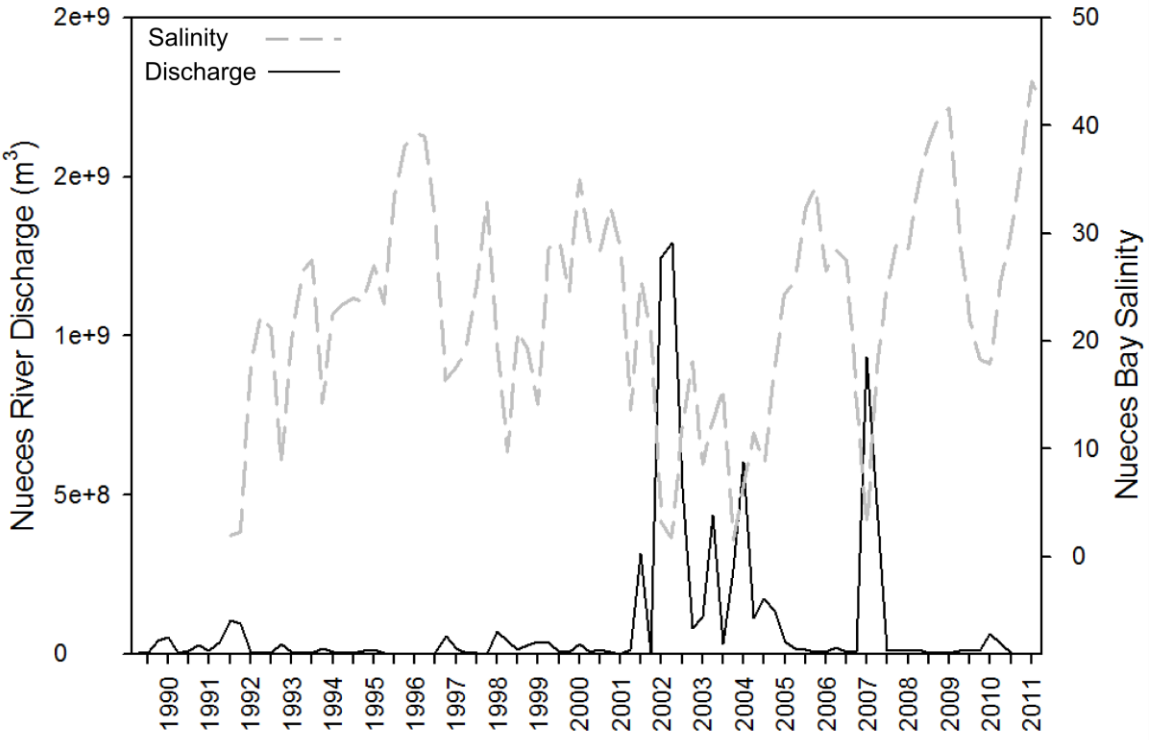

Figure 2.2: Long term trends in Nueces River discharge and Nueces Bay salinity (1990-2011).

VEGETATION AND POREWATER MONITORING

The abundance and distribution of emergent plants was monitored quarterly over a twelve year period from 1999 to 2011 at three sites in the low marsh. The resulting dataset documents observed changes in seasonal plant community composition and coverage in response to changes in soil porewater characteristics. The abundance of emergent plants for this time period was estimated from percent cover data collected within 0.25 m^2 quadrats (percent cover data was used as a proxy for abundance). Measurements were taken at 2-m intervals along 6 parallel 10 m transects (30 quadrats / site) at each of the three sites (Figure 2.1). Soil characteristics were obtained by extracting water from soil cores (2.5 cm diameter x 10 cm length) by centrifugation. The extracted water was analyzed for salinity using a handheld refractometer (Reichert Scientific Instruments, Buffalo, NY) and porewater ammonium (NH_4+) using standard colorimetric techniques [*Parsons et al.* 1984]. Separate soil cores were collected for determination of soil moisture. Cores were transferred to the laboratory in sealed containers and dried to a constant weight in a 60 °C oven. The soil moisture content was calculated as the change in weight following drying and standardized to initial wet weight. Variations in porewater salinity and corresponding vegetation characteristics were evaluated with respect to freshwater inflow [*USGS* 2011].

IDENTIFICATION OF INDICATOR SPECIES

Several vegetation species were evaluated as potential indicators of ecosystem condition, including *B. frutesecens*, *S. virginica*, and *S. alterniflora*. Indicator species were ultimately selected based upon documentation of their sensitivity to stress, ease of assessment, and known population distribution [*Dale and Beyeler* 2001]. Although all three species were evaluated, we thought *S. virginica* and *B. frutescens* were unlikely candidates because *S. virginica* is relatively insensitive to salinity stress [*Forbes et al.* 2008; *Rasser* 2009] and *B. frutescens* is primarily found at high marsh elevations. Literature surveys and preliminary analysis suggested that *S. alterniflora* was a strong indicator species candidate [*Webb* 1983]. The salinity tolerance of indicator species was estimated by comparing vegetation abundance data against corresponding porewater salinity measurements. Determination of freshwater inflow needs was calculated from the relationship between freshwater inflow and porewater salinity targets modeled as an exponential decay function.

STATISTICAL ANALYSES

The relationship between vegetation cover and environmental variables was examined using canonical correspondence analysis (CCA). CCA was performed on a species-environment matrix that included quarterly measurements of vegetation distribution (grouped according to transect) versus corresponding averages of environmental variables (porewater salinity, porewater NH_4+, soil moisture, and distance to nearest tidal creek). Vegetation cover data was left unstandardized in order to retain information on the species-environment relationship [*Kenkel* 2006]. Non-metric multidimensional scaling (NMDS), based on a Bray-Curtis similarity index, was used to

evaluate changes in the vegetation community with respect to hydroclimatic periods. Vegetation data was log (x +1) transformed prior to NMDS and CCA in order to normalize the data. The salinity tolerance of potential indicator species was evaluated using field observations. All statistical analyses were carried out in the R statistical program (version 2.11.1). Both CCA and NMDS analyses were carried out using the vegan package [*Okansen* et al. 2007].

Results

CLIMATE AND HYDROLOGY

Freshwater inflows to the Nueces Estuary exhibited significant variation throughout the study period and were characterized by distinct wet and dry periods (Figure 2.3). There were three periods with measurable freshwater inflow in 2002-2004, 2007, and 2010. These relatively wet periods were preceded by extended drought periods in 1999-2001, 2005-2006, and 2008-2009 (Figure 2.3). The end of the study period in 2011 was characterized by an exceptional drought period (see National Climate Data Center, http://www.ncdc.noaa.gov/sotc/drought/2011/). Average annual freshwater inflow to the Nueces Estuary was 5.57×10^8 m^3y^{-1} over the course of the study period (Table 2.2). Porewater salinity was lower during wet periods when large freshwater inflow events flushed soils of accumulated salts (Figure 2.4). During drought periods and in the absence of freshwater inflow, porewater salinity was often elevated to values several times that of standard seawater (Figure 2.4). Porewater salinity was nearly equivalent to the salinity of nearby tidal creeks (Figure 2.5).

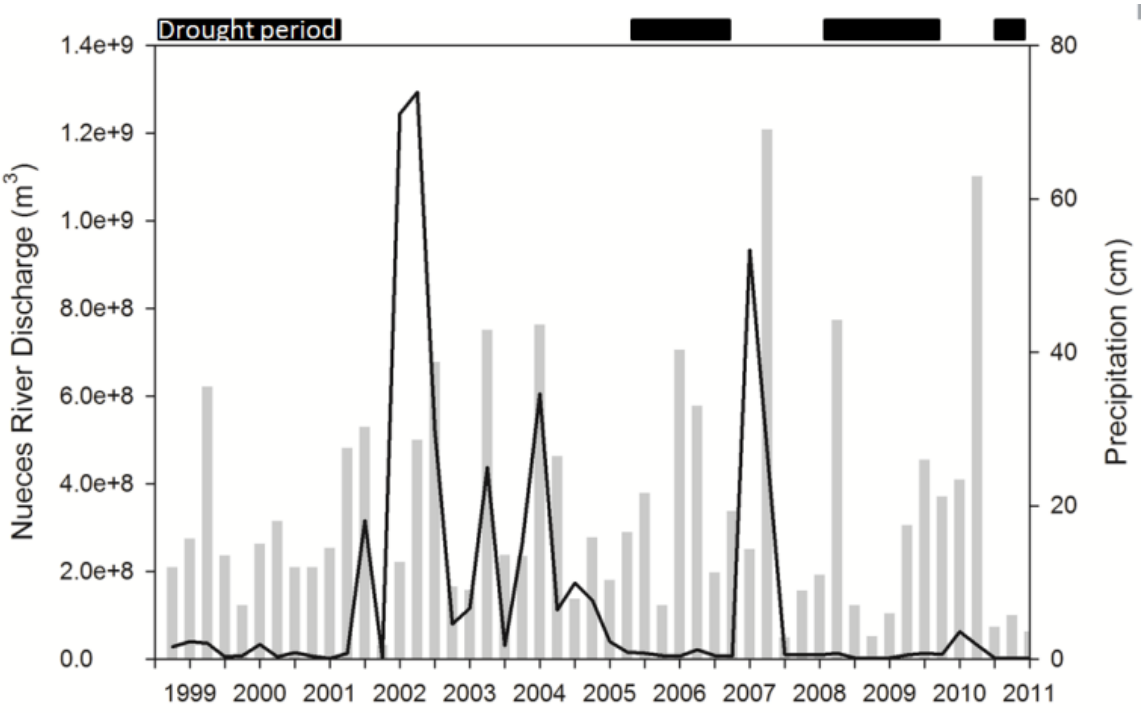

Figure 2.3: Quarterly precipitation (shaded bars) at the Corpus Christi airport and freshwater inflow (solid line) to the Nueces Estuary via the Nueces River (1999-2011). Four drought periods in 1999-2001, 2005-2006, 2008-2009, and 2011 are identified by shaded boxes.

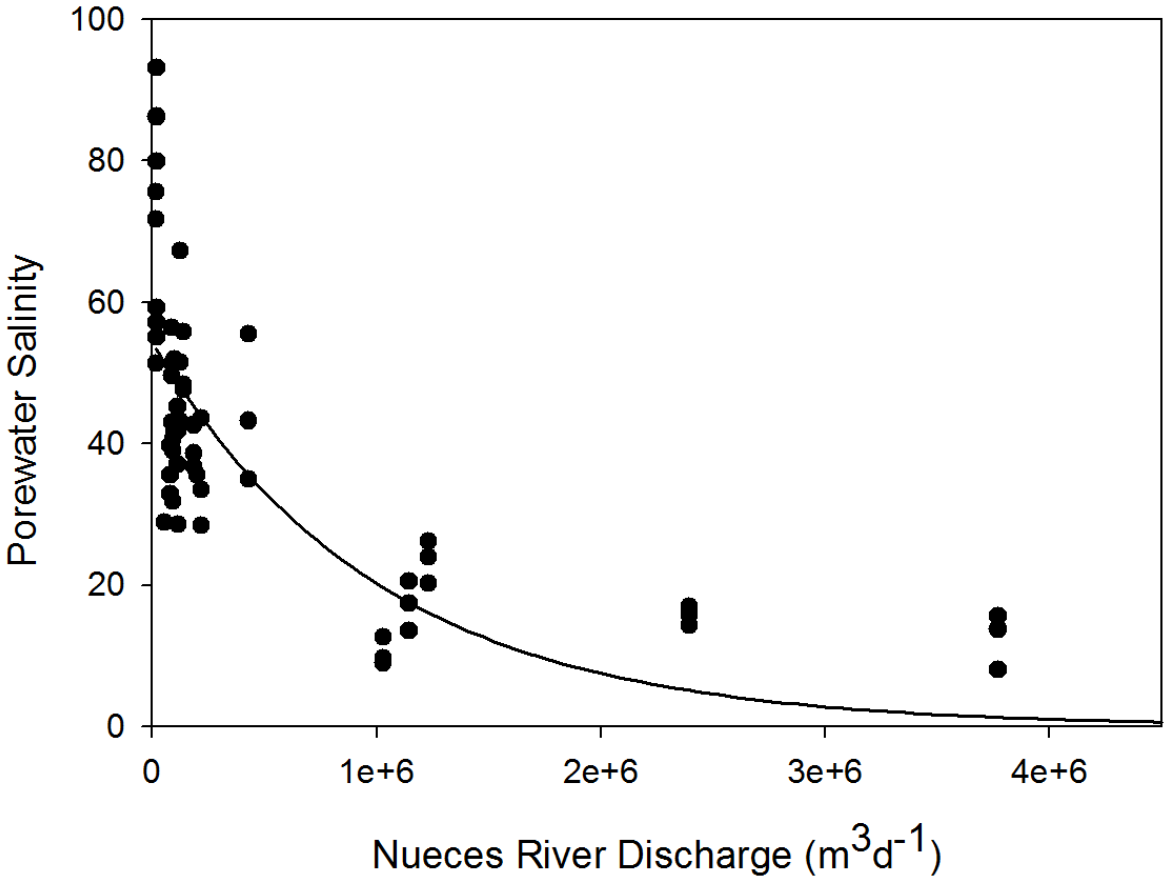

Figure 2.4: Relationship between freshwater inflow (Nueces River: USGS #08211500) and porewater salinity along the creek bank in the low marsh. Regression curve is a best fit line for an exponential decay function ($y = 54.39\ e^{(-9.89e-7)x}$, $R^2 = 0.63$). A salinity target of 25 yields a freshwater inflow target of approximately $7.86 \times 10^5\ m^3d^{-1}$.

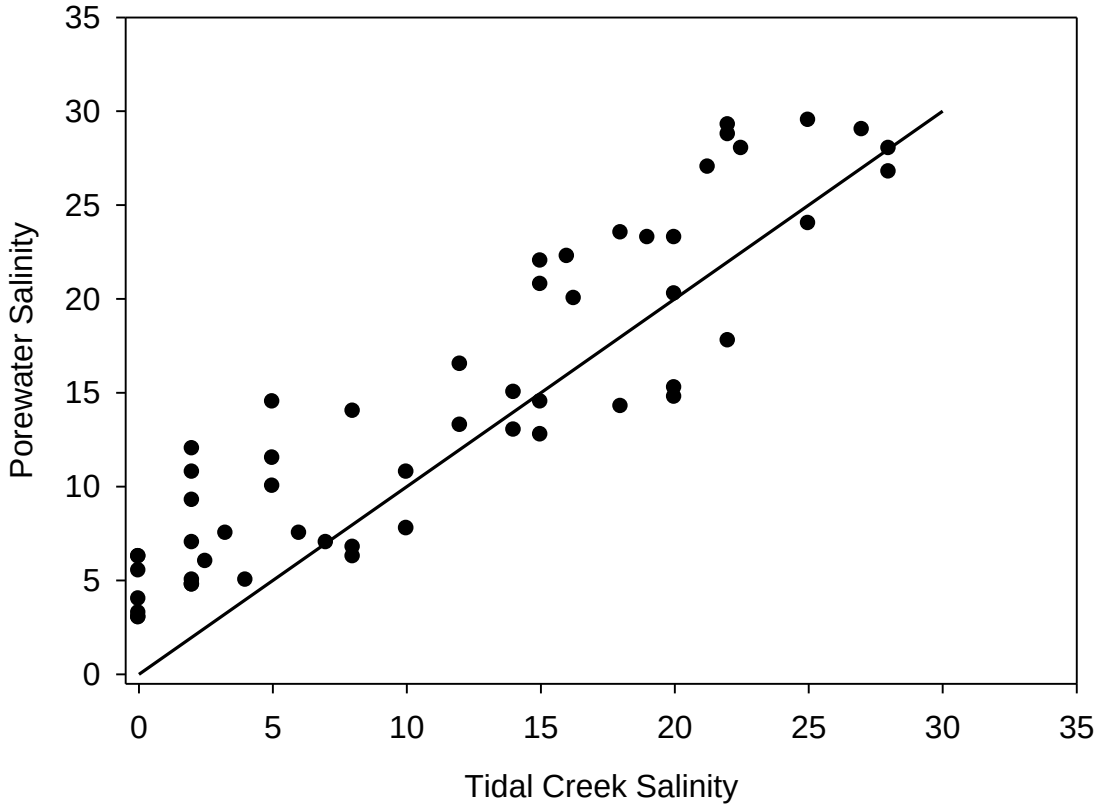

Figure 2.5: Corresponding measurements of creekbank porewater and tidal creek salinity in relation to their theoretical one-to-one relationship (solid line).

HYDROLOGIC IMPACTS ON EMERGENT PLANTS

Hydrology clearly influenced that plant community of the Nueces River Delta (Figure 2.6, 2.7). The first two CCA ordination axes explained 92% of the variance for emergent plant cover (Table 2.1). However, the first axis had considerably greater explanatory power (77.9%) than the second axis (14.1%). The first axis was negatively correlated with soil moisture and positively correlated with porewater salinity. This suggests that species' habitat is separated primarily according to soil moisture and porewater salinity (Table 2.1). While *S. alterniflora* cover was most common in brackish water-logged sediments, *B. frutescens* cover dominated well-drained saline sediments (Figure 2.9). The composition of vegetation communities immediately following major freshwater inflow events was highly variable (Figure 2.6). However, *Spartina alterniflora* was consistently more abundant following freshwater inflow events. Vegetation communities during drought periods were characterized by an abundance of *Salicornia*

Table 2.1: Results of Canonical Correspondence Analysis.

Constraining Variables	**Axis 1**	**Axis 2**
Porewater Salinity	0.59	-0.45
Porewater Ammonium	-0.01	0.34
Soil Moisture	-0.94	0.27
Distance to Tidal Creek	0.40	0.37
Distance to Nueces Bay	0.59	0.64
% Variance Explained	**77.93**	**14.08**

virginica. Analysis of percent cover data provided evidence of a distinct vegetation assemblage corresponding with identified drought periods (Figure 2.6). We used non-metric multidimensional scaling of emergent plants according to site and time period in order to test if this vegetation assemblage is unique to drought periods (Figure 2.7). We found a distinct clustering according to the hydroclimatic periods identified in Figure 2.3. For example, almost all (94%) of drought period assemblages at site 254 fell within the same similarity envelope (Figure 2.7). Likewise, drought period assemblages at site 450 and 270 were also found within the same similarity envelope (73% and 38% respectively). The lack of clustering at site 270 can be attributed to massive disturbance caused by a flooding event in 2002. This flood event eroded almost 4 m from the creekbank and permanently changed the community from a mixed vegetation assemblage to one dominated primarily by *Borrichia frutescens* (Dunton, unpublished data). As a result, early drought assemblages (1999-2002) at this site are not comparable to post-flood assemblages.

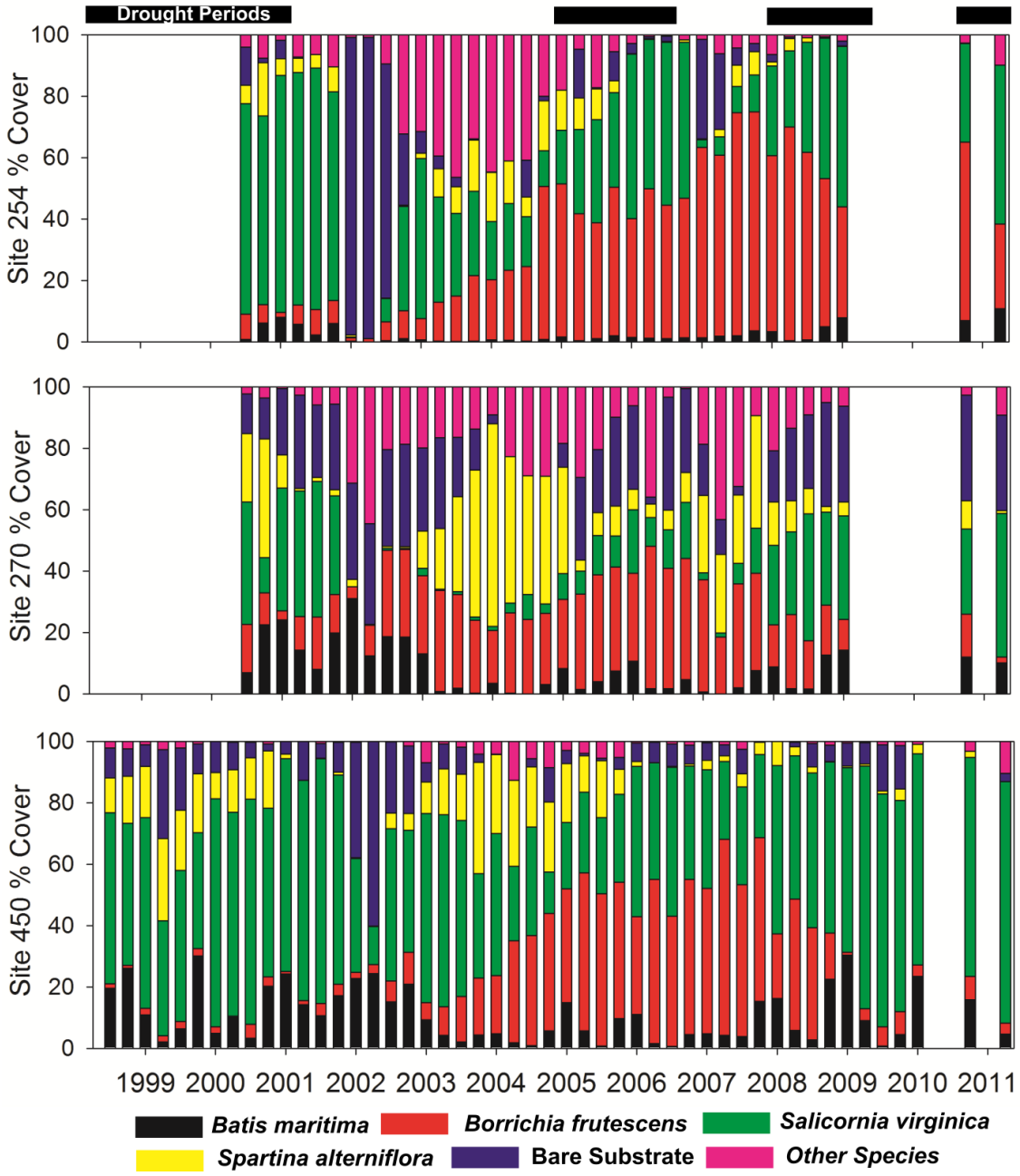

Figure 2.6: Quarterly percent cover of emergent plants at selected sites in the Nueces River Delta for the period 1999-2011. Shaded boxes at top indicate the occurrence of drought periods. Drought periods were defined as years with inflows below the median.

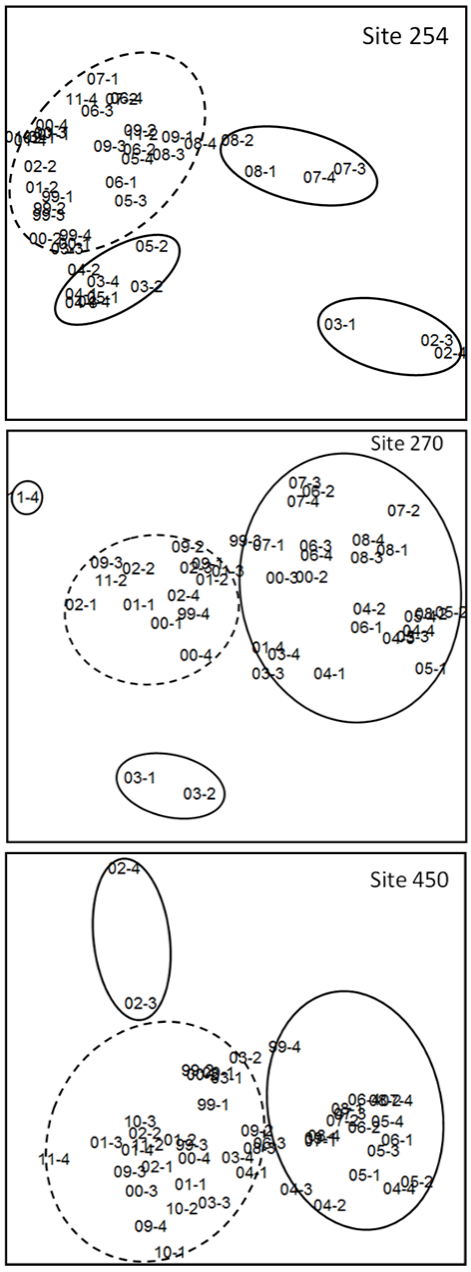

Figure 2.7: Non-metric multi-dimensional scaling ordination of emergent plant communities averaged by station and quarterly sampling date formatted as YY-Quarter. For example, Winter 2000 is denoted by 00-1. Similarity clusters are defined at 60% similarity by the Bray-Curtis method. Clusters are outlined to show corresponding drought period (dashed circles) and wet period (nondashed circles) vegetation assemblages.

ESTIMATION OF FRESHWATER INFLOW REQUIREMENTS

The salinity tolerance of potential indicator species was determined for *S. alterniflora*, *B. frutescens*, and S. *virginica* based on changes in percent cover in relation to porewater salinity (Figure 2.8, 2.9). The abundance of *S. alterniflora* fluctuated from a minimum cover near 0% (Spring 2009) to a maximum cover of approximately 66% (Summer 2004, Figure 2.8). Spatial variations in *S. alterniflora* cover were evident among study sites. The site with the highest cover, site 270, is close to Nueces Bay and has the lowest topographic relief. In contrast, the site with the lowest maximum cover, site 254, has a pronounced creekbank levee [*Rasser* 2009]. Observed spatial patterns among sites were consistent with the idea that cover *S. alterniflora* is limited to regularly flushed low elevation areas. Consistent with our hypothesis, fluctuations in *S. alterniflora* cover were clearly related to porewater salinity and freshwater inflow. Porewater salinities exceeding 25 resulted in dramatic declines in *S. alterniflora* coverage (Figure 2.8, 2.9). There were only two outliers where *S. alterniflora* coverage was substantial (>25%) and salinity exceeded 25. These outliers were associated with the lagged response of plants to rapid increases in salinity during the onset of drought in 2005. Although freshwater inflows were concentrated in the summer season, there was no consistent relationship between time of year (season) and standing coverage of *S. alterniflora*. However, increases in cover from one season to the next occurred primarily (74%) during the spring and summer rather than during fall and winter (26%, Figure 2.8).

The observed relationship between porewater salinity and freshwater inflow was investigated with respect to S. *alterniflora* abundance. An exponential decay fit to this relationship provided a means to estimate freshwater inflow corresponding to a given salinity target. This study determined that achieving a porewater salinity target of 25 requires a Nueces River discharge of approximately 2.87 x 10^8 m^3y^{-1} (Table 2.2). However, this value can be expressed as a range between 2.2 and 3.7 x 10^8 m^3y^{-1} owing to variations in published salinity tolerance values for *S. alterniflora* between 20 - 30 [*Webb* 1983; *Bertness* 1991].

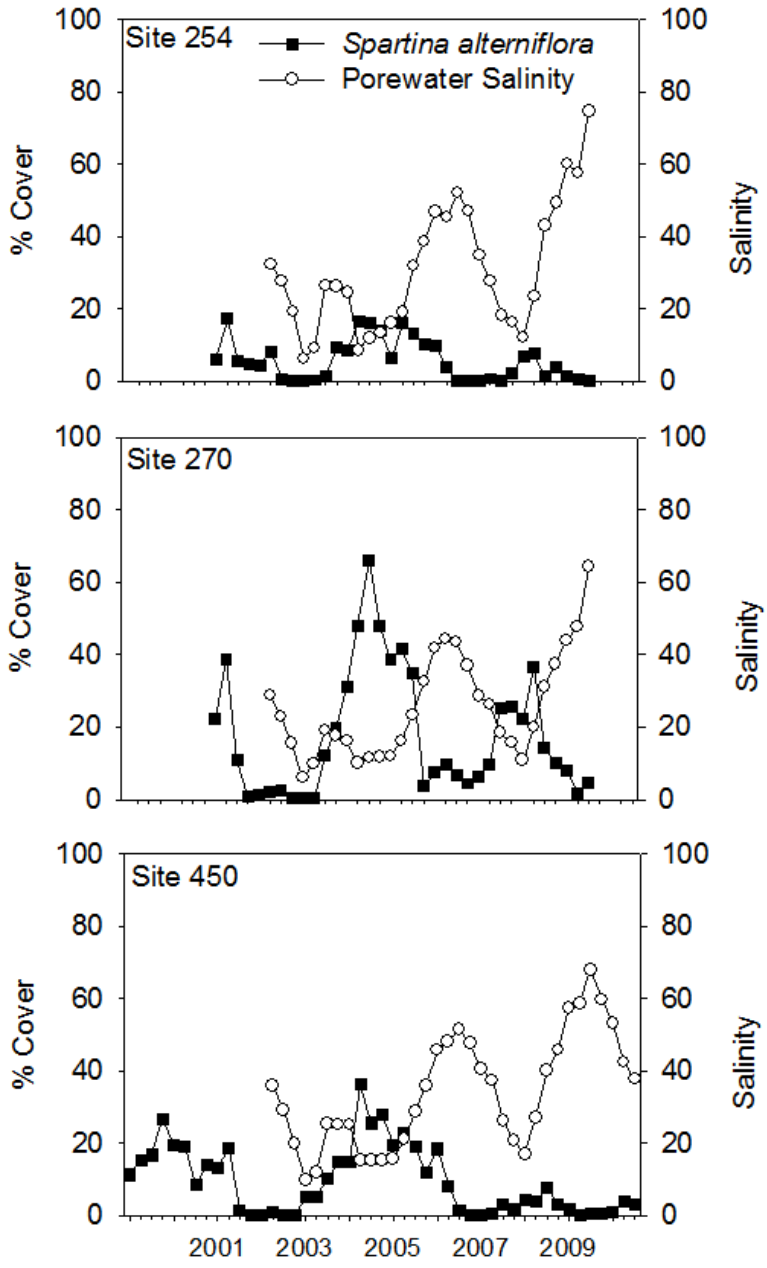

Figure 2.8: Porewater salinity (white circles) and percent cover of *Spartina alterniflora* (black squares) along the creek bank in the low marsh. Porewater salinities exceeding 25 result in declines of *S. alterniflora* abundance.

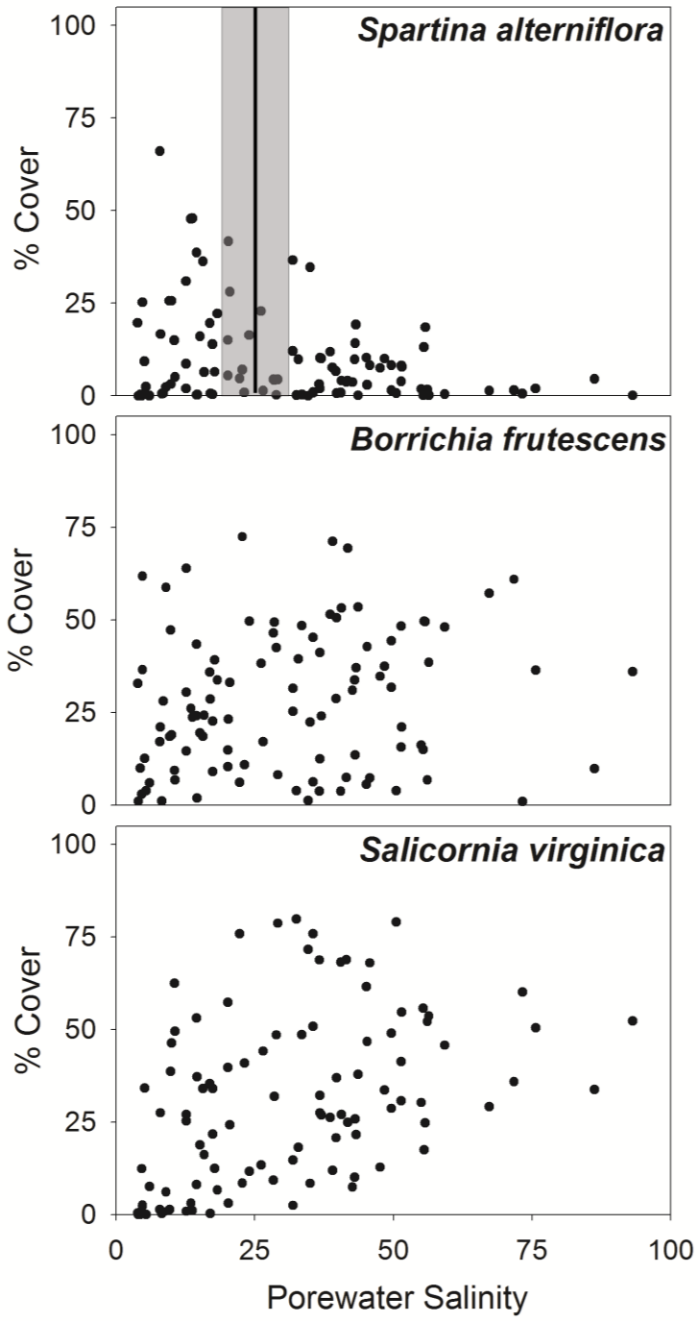

Figure 2.9: Percent cover of individual plant species (*S. alterniflora*, *B. frutescens*, and *S. virginica*) relative to variations in porewater salinity. The salinity tolerance (shaded box) of *S. alterniflora*, estimated at 25 ± 5, was estimated from empirical measurements and published literature values [*Webb* 1983; *Bertness* 1991].

Discussion

VEGETATION RESPONSE TO FRESHWATER INFLOW

Freshwater inflow events impact the Nueces Estuary by flushing salts, delivering nutrients, and distributing sediments [*BOR* 2000]. The most dramatic of these effects is the flushing of salts following large magnitude freshwater inflow events. For example, flooding in 2002 caused extensive freshening of Nueces Bay, dropping salinity values near standard seawater to values typical of freshwater and brackish systems (Figure 2.2). Two approaches have been used to assess the impact of freshwater inflow events on the Nueces Estuary. Early studies focused on the impact of individual hydrographic events in order to define flooding thresholds and flow regimes [*Ward et al.* 2002]. Later studies aggregated hydrographic events into distinct hydroclimatic periods [*Forbes and Dunton* 2006; *Montagna et al.* 2009]. In this paper we take the latter approach in order to examine how long term (>10 years) variations in freshwater inflow impact the emergent plant community of the Nueces Delta.

Previous studies have shown that the emergent plant community is responsive to variations in salinity and freshwater inflow [*BOR* 2000; *Alexander and Dunton* 2002; *Forbes and Dunton* 2006]. However, this study is unique in that it considers both the wettest period (2002 -2004) and the driest period (2008-2009) since reservoir construction. Our results demonstrate that the vegetation community typical of drought periods is distinct from that of wet periods (Figure 2.7). In addition, communities observed during early droughts (1999-2001) reappeared during subsequent dry periods in

2005 and 2008 (Figure 2.6, 2.7). These drought period communities were characterized by a high abundance of *S. virginica* and a low abundance of *S. alterniflora* (Figure 2.6). The time required for the reappearance of drought period assemblages was related to the magnitude of freshwater inflow events during the preceding wet period. High freshwater inflows during 2002-2004, the wettest period during this study, extended the time period between the reemergence of drought period vegetation communities. Furthermore, vegetation communities returned to a drought assemblage after only one year following moderate inflows in 2007 (Figure 2.6, 2.7). Our results are consistent with previous studies regarding the response of the plant community to salinity and freshwater inflow (Forbes and Dunton 2006).

One way in which environmental stress is expressed in the vegetation community is through zonation. Zonation is characterized by distinct banding or spatial separation of species depending on differing tolerance to environmental stress and interspecific competition for resources [*Adams* 1963; *Pennings et al.* 2005]. Typically, this occurs in response to variations in inundation frequency corresponding with an elevation gradient [*Rasser* 2009]. Zonation can be observed in the Nueces Delta under intermediate flooding disturbance. However, during extreme drought or flooding, zonation bands are dissolved and extensive bare areas are created [*Alexander and Dunton* 2002]. Large magnitude events, such as floods, are known to cause wholesale reorganization of the vegetation community [*Forbes and Dunton* 2006]. The NMDS analysis from this study confirms a consistent reorganization of the plant community following flood disturbances (Figure 2.7).

This finding is important because the use of emergent vegetation as indicators of ecosystem condition is predicated on the assumption that community structure is predictable under a given set of hydroclimatic conditions. Vegetation communities, in this study, followed a predictable trajectory. First, bare areas were created following large inflow events and were initially colonized by stress intolerant species such as *S. alterniflora* and *Sueda maritima*. Next, in the absence of freshwater inflow, these individuals were eventually replaced by the moderately stress tolerant *B. frutescens*. Finally, the onset of drought conditions encouraged the replacement of all other species by the stress tolerant *S. virginica* (Figure 2.9). Our observation that *S. virginica* abundance increases during drought periods is consistent with a study by Forbes and Dunton (2006) that demonstrated the displacement *S. virginica* by *B. frutescens* following freshwater inflow events. In addition, a variety of studies determined that *S. virginica* is resilient to extreme environmental stress [*Zedler* 1983; *Forbes and Dunton* 2006; *Rasser* 2009). Our results demonstrate that frequent freshwater inflow events are required for the maintenance of a *S. alterniflora* creekbank habitat.

MANAGEMENT OF FRESHWATER INFLOWS

Reduced freshwater inflows, due to extreme drought and reservoir construction, prompted legislative mandates calling for ecological assessments of Texas estuaries with the purpose of determining freshwater inflow requirements [*TDWR* 1982; *BBEST* 2011]. Although studies have utilized numerous methods to derive hydrologic data output, nearly all ecological studies concerning freshwater inflow have used the physiological (salinity) tolerance of indicator species to set inflow bounds [*Powell et al.* 2002]. These

tolerances are typically determined from a suite of environmentally sensitive or economically important indicator species. As a result, effective management and allocation of freshwater for the ecological benefit of estuarine wetlands requires detailed knowledge of the physiological tolerances of resident organisms. In many cases, these are estimated from limited and expensive field surveys. This study demonstrates that emergent marsh plants respond predictably to environmental conditions and provide valuable information regarding the ecological condition of estuaries. Salt marsh plants are valued as a substrate stabilizer, a contributor to food webs, and a refuge for a variety of nekton species including fish, invertebrates, and migratory birds [*Henley and Rauschuber* 1981; *Zedler and Kercher* 2005]. Given that they are also inexpensive to survey, they make ideal candidates to be included in a suite of indicator species.

S. alterniflora is the only emergent plant species considered in this study that consistently reflects environmental conditions in tidal creeks (Figure 2.8, 9), exhibits a salinity tolerance similar to other faunal estuarine indicator species [*BBEST* 2011], and provides an ecologically important habitat [*Kneib* 2003]. Our results clearly show that the coverage of *S. alterniflora* is regulated by porewater salinity. For instance, the cover of this species was substantially reduced at salinities exceeding 25, and this finding is consistent with previous studies investigating the salinity tolerance of *S. alterniflora*. A study by Webb (1983) found that porewater salinities exceeding 25 resulted in significant reductions in density, height, and standing biomass. Integrative studies by BBEST (2011) and TDWR (1982) illustrated that the freshwater inflow needs of *S. alterniflora* is nearly identical to that of other common indicator species such as the blue crab (*Callinectes*

sapidus), Atlantic croaker (*Micopogonias undulates*), and eastern oyster (*Crassostrea virginica*). Therefore, one would expect that the abundance of *S. alterniflora* serves as a reasonable proxy for the abundance of these higher trophic level organisms.

S. alterniflora stands represent a unique habitat because it is the only species found at the lowest exposed elevations in the Nueces Delta. Cover of this species is limited to the areas directly adjacent to creekbanks that fall within the range of daily tidal variation [*Rasser* 2009]. Under stressful environmental conditions cover of *S.alterniflora* is converted to open water habitat. This conversion represents the loss of a unique habitat as *S. alterniflora* is known to promote nekton density and production [*Whaley and Minello* 2002; *Kneib* 2003]. In the Nueces Delta, the benefits of *S. alterniflora* cover to higher trophic level organisms likely occur indirectly through the provision of habitat rather than direct carbon assimilation [*Wallace* 2011].

Several previous studies have surveyed the distribution of *S. alterniflora* in the Nueces Delta. A study by Forbes and Dunton (2006) found that *S.alterniflora* could be classified as a clonal dominant. Under favorable conditions, this species increased in abundance and restricted the distribution of other emergent plant species. Additional surveys by Forbes and Dunton (2006) and Rasser (2009) found that *S.alterniflora* abundance was concentrated at the edge of tidal creeks and was generally lower than other plant species. However, both of these studies focused on high marsh habitats (further from Nueces Bay) beyond a railroad track that bisects the Nueces Delta (Figure 2.1). This railroad crossing restricts hydraulic flow between the low and high marsh and substantially dampens tidal fluctuations [*Ryan* 2011]. *S. alterniflora* is likely not found in

these areas because it depends on frequent inundation. In the low marsh (closer to Nueces Bay), *S. alterniflora* is found in much greater abundance during wet years (Figure 2.6).

Although numerous studies have examined the freshwater inflow needs of the Nueces Estuary, no study has yet produced a comprehensive comparison of inflow estimates from diverse methodologies and time periods. Previous estimates of freshwater inflow needs in the Nueces Delta vary widely from annual inflows of only 1.12×10^8 to 4.98×10^8 m^3y^{-1} (Table 2.2). Estimated freshwater inflow needs have varied among studies because of historical reservoir development, differing analytical methods, and time scales. Early studies estimating freshwater inflow requirements of the Nueces Estuary, prior to reservoir development, determined that adequate ecosystem function is achieved at annual inflows of 4.98×10^8 m^3y^{-1} [*TDWR* 1982]. Subsequent estimates following reservoir construction are much lower (1.71×10^8 m^3y^{-1}, *Pulich et al.* 2002; 1.12×10^8 m^3y^{-1}, *BOR* 2000). A study by BOR (1975), predating reservoir construction, determined that average annual inflows from 1972 to 1975 were 5.07×10^8. This is well above the average annual inflows observed throughout this study period and clearly not realistic given increasing municipal water demand and upstream reservoir construction. However, more recent estimates by Pulich et al. (2002) and Bureau of Reclamation (2000) barely exceed median observed inflows and may underestimate actual inflow needs (Table 2.2). Our estimate, based on the abundance of *S. alterniflora,* falls between historically high estimates and recent low estimates at a conservative 3.13×10^8 m^3y^{-1} (Table 2.2). Achieving such annual freshwater inflows requires less than the average

annual inflow observed during the study period (Table 2.2). Although Nueces River flows exceeded our estimated freshwater inflow requirements in 5 of the 11 years encompassed by this study (1999-2011), they only exceeded this target 6 years between 1990 and 2011 (Figure 2.2).

Table 2.2: Gauged freshwater inflows to the Nueces Estuary via the Nueces River (USGS gauge #08211500) and estimated annual freshwater inflow needs. Numbers for this study were calculated based on historical attainment of a 25 porewater salinity target for vigorous *Spartina alterniflora* growth in the Nueces Delta. Estimates are reported as the average or median inflow observed or estimated among specified years.

Inflow Type	**Sampling Method**	**Freshwater Inflow (m^3y^{-1})**	**Date Range**	**Source**
Gauged	Average	$7.87x10^7$	1990-1998	This Study
	Average	$5.57x10^8$	1999-2011	
	Median	$1.18x10^8$	1999-2011	
Estimated Need	Average	$4.98x10^8$	1962-1976	TDWR 1982
	Average	$1.12x10^8$	1995-present	BOR 2000
	Average	$1.71x10^8$	1978-1997	Pulich et al. 2002
	Range	$2.20 – 3.69x10^8$	1999-2011	This Study

We primarily focused on annual and interannual patterns in freshwater inflow because it is consistent with municipal water management strategies discussed in previous studies. However, there are likely important variations in freshwater inflow on time scales not explicitly considered in our analyses. Analysis of historic freshwater inflow patterns suggests that decadal variations in freshwater inflow may be occurring [*BOR* 2000]. For example, while observed annual freshwater inflows regularly exceeded our inflow requirement estimates listed in Table 2.2, they were met in only one year between 1990 and 2000 [*USGS* 2011]. It is also likely that seasonal inflow patterns are important given that seedling germination mostly occurs in the spring [*Alexander and Dunton* 2002]. Furthermore, previous studies have found evidence that seedlings exhibit different physiological tolerances to environmental stress than adult plants [*Shumway and Bertness* 1992]. In the Nueces Delta, Alexander and Dunton (2002) found that seed germination and expansion of *Salicornia bigelovii* was facilitated by freshwater input. Water managers tasked with resolving conflicts between municipal use and ecological benefits should consider altering the timing of freshwater inflows to coincide with critical germination periods of *S. alterniflora.* Future research should assess the impacts of freshwater inflow timing on *S. alterniflora* abundance. Greater knowledge of the importance of inflow timing is required before our recommendations can be applied in a management context.

FUTURE IMPACTS

The overall extent of emergent salt marsh plants in the Nueces Delta is likely to shrink as a result of continued decreases in freshwater inflow concurrent with more

erratic and possibly decreasing precipitation due to global climate change [*Forbes and Dunton* 2006]. Ward and Valdes (1995) evaluated the impact of global climate change on Texas water resources relative to a scenario characterized by a 2 °C increase in temperature and a 5% decrease in precipitation. Based on this scenario, Ward and Valdes (1995) projected a 35% decrease in freshwater inflow to Texas estuaries. Our results suggest that if droughts become longer and more frequent, *S. virginica* will likely replace *S. alterniflora* and make up a greater proportion of the overall community. This has important implications for the ecological health of the Nueces Delta and provision of ecosystem services. Since the rooting depth of *S. virginica* is much shallower than the rooting depth of *S. alterniflora*, this shift would decrease the ability of vegetation to provide sediment stabilization. Changes in the plant community of the Nueces Delta may provide a forecast of future changes in wetter, more northerly estuaries [*Kirwan et al.* 2009]. Future monitoring efforts in these estuaries should focus on northward latitudinal shifts in *S. alterniflora* in response to freshwater inflow and global climate change.

Appendix

NUECES DELTA LONG TERM MONITORING DATA

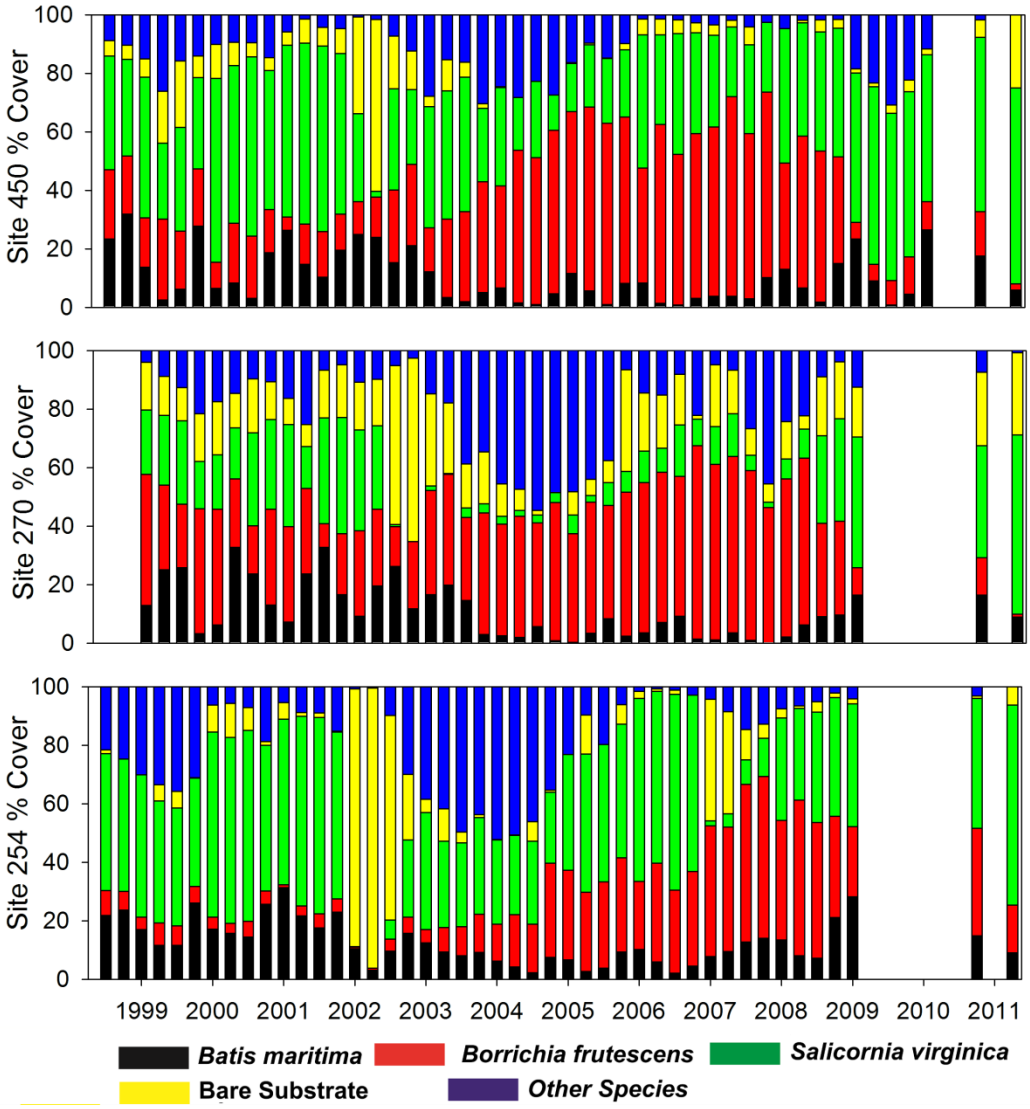

Figure A1: Quarterly percent cover of emergent plants at selected sites (450, 270, 254) in the Nueces River Delta for the period 1999-2011. Sites are ordered top-bottom at increasing distance from Nueces Bay.

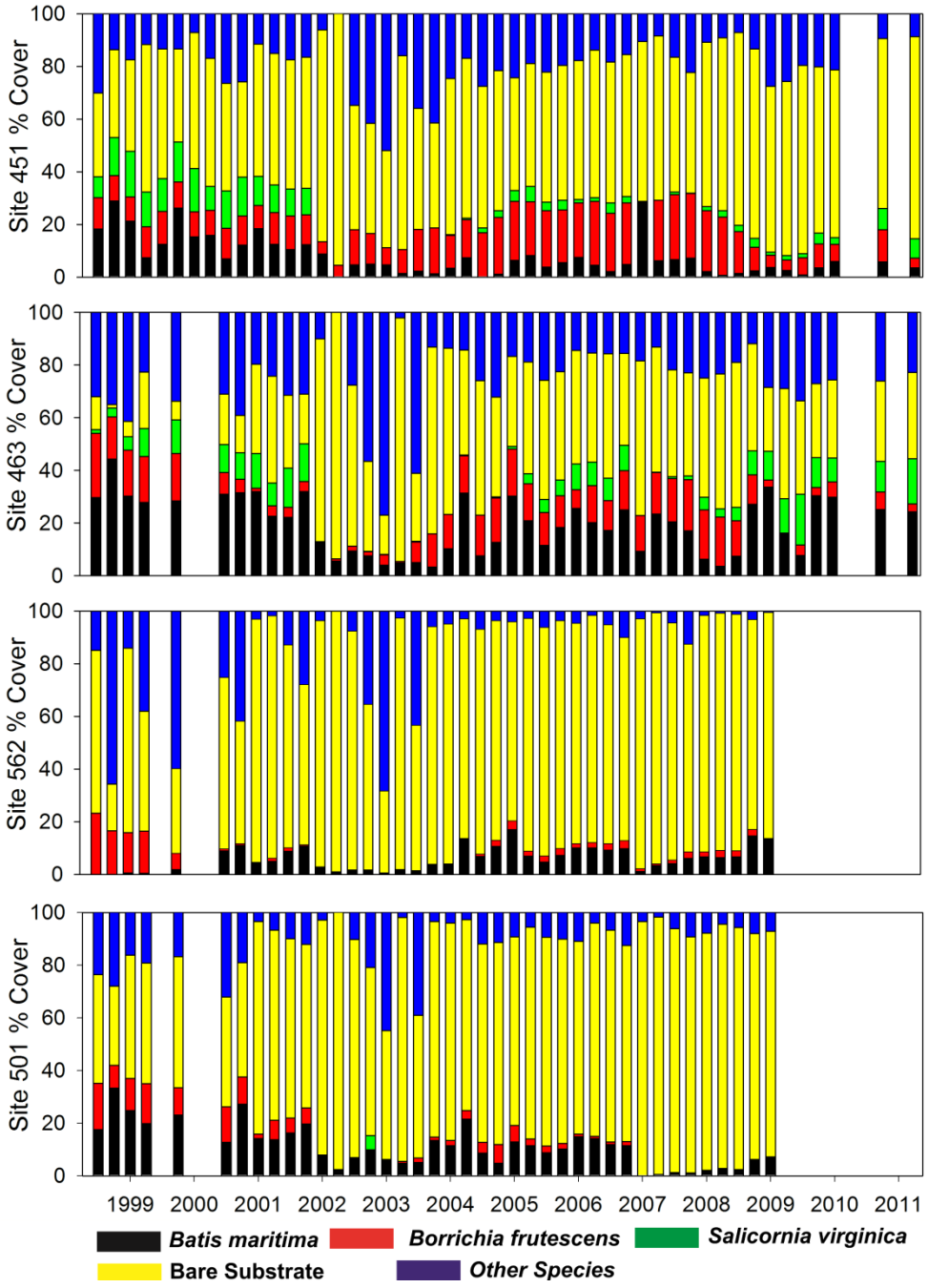

Figure A2: Quarterly percent cover of emergent plants at selected sites (451, 463, 562, 501) in the Nueces River Delta for the period 1999-2011. Sites are ordered top-bottom at increasing distance from Nueces Bay.

POREWATER SALINITY MASS BALANCE SCRIPT

```
'Visual Basic Script for simulating mass balance of salt
and water in sediment porewaters
Public Sub PwBallance()

'Variable assignments
     Dim n As Long
     Dim i As Long
     Dim sglPrecipitation As Single 'mm'
     Dim sglEvaporation As Single 'mm'
     Dim sglPrevSalinity As Single 'concentration'
     Dim sglpwsalinity As Single
     Dim sglWaterVolume As Single
     Dim sglTcSalinity as Single
     Dim lngStart As Long
     Dim lngEnd As Long
     Dim rngResults As Range
     Dim rngDates As Range
     Dim rngSalinity As Range
     Dim boolPrecip As Boolean
     Dim boolInnun As Boolean
     Dim boolPrevInnun As Boolean
     Dim intPrevPrecipDay As Integer
     Dim intPrevInnunDay As Integer
     Dim intSinceInnunDay As Integer
     Dim intFieldCapacity As Integer
     'average time(days) to field capacity
     Dim intFieldCapacityVolume As Integer
     'volume of pedon at field capacity
     Dim intDrainage As Integer
     'average time(days)water is impounded after inundation
     Dim sglSalEvapCoefficient As Single
     Dim sglInnunVol As Single
     Dim sglCsBudget As Single
     Dim sglCsLeached As Single
     Dim sglDensity As Single
     Dim sglRe As Single
     Dim sglK1 As Single
     Dim dblR As Double
     Dim sglC1Hat As Single
     Dim sglWaterSurfaceArea As Single
     Dim sglPwSaltMass As Single
     Dim sglCsBudgetMass As Single
     Dim sglDrainWater As Single
```

```
     Dim sglInnunFreq As Single

     i = 1

'Set initial values (constants)
     lngStart = Range("Start").Value
     lngEnd = Range("End").Value
     sglpwsalinity = 24
     sglWaterVolume = 136

     'mm3 assuming porosity of 0.68 experimentally
     determined'
     Worksheets("TimeSeries").Cells(1, 3) = sglpwsalinity
     boolPrevInnun = True
     sglWaterSurfaceArea = 200
     'mm assuming an ideal pedon 0.2 m deep
     sglPwSaltMass = sglpwsalinity * (1.02 *
          sglWaterVolume)

'Clear out the results table
     Set rngResults =
     Range(Worksheets("TimeSeries").Cells(2, 2),
     Worksheets("TimeSeries").Cells(65536, 16))
     rngResults.Clear

'Start loop
     For n = 1 To (lngEnd - lngStart + 1)

'Locate variables for the current day'

     sglTcSalinity =
          Worksheets("Innundation").Application.WorksheetFu
          nction.VLookup((lngStart + n - 1), _
               Range(Worksheets("Innundation").Cells(2, 1),
          Worksheets("Innundation").Cells(65536, 2)), 2,
          False)
     boolInnun =
          Worksheets("Innundation").Application.WorksheetFu
          nction.VLookup((lngStart + n - 1), _
                 Range(Worksheets("Innundation").Cells(2,
          1), Worksheets("Innundation").Cells(65536, 3)),
          3, False)
```

```
sglPrecipitation =
     Worksheets("Precipitation").Application.Worksheet
     Function.VLookup((lngStart + n - 1), _
            Range(Worksheets("Precipitation").Cells(2,
     1), Worksheets("Precipitation").Cells(65536, 3)),
     2, False)
intPrevPrecipDay =
     Worksheets("Precipitation").Application.Worksheet
     Function.VLookup((lngStart + n - 1), _
            Range(Worksheets("Precipitation").Cells(2,
     1), Worksheets("Precipitation").Cells(65536, 3)),
     3, False)
sglEvaporation =
     Worksheets("Precipitation").Application.Worksheet
     Function.VLookup((lngStart + n - 1), _
            Range(Worksheets("Precipitation").Cells(2,
     1), Worksheets("Precipitation").Cells(65536, 5)),
     5, False)
sglInnunVol =
     Worksheets("Innundation").Application.WorksheetFu
     nction.VLookup((lngStart + n - 1), _
            Range(Worksheets("Innundation").Cells(2,
     1), Worksheets("Innundation").Cells(65536, 4)),
     4, False)
intPrevInnunDay =
     Worksheets("Innundation").Application.WorksheetFu
     nction.VLookup((lngStart + n - 1), _
            Range(Worksheets("Innundation").Cells(2,
     1), Worksheets("Innundation").Cells(65536, 7)),
     7, False)
intSinceInnunDay =
     Worksheets("Innundation").Application.WorksheetFu
     nction.VLookup((lngStart + n - 1), _
            Range(Worksheets("Innundation").Cells(2,
     1), Worksheets("Innundation").Cells(65536, 8)),
     8, False)
sglInnunFreq =
     Worksheets("Innundation").Application.WorksheetFu
     nction.VLookup((lngStart + n - 1), _
            Range(Worksheets("Innundation").Cells(2,
     1), Worksheets("Innundation").Cells(65536, 6)),
     6, False)

If n = lngStart Then
```

```
          sglPrevSalinity = sglpwsalinity
     End If

'Transfer water into the pedon (add water from
innundation)'
     If sglInnunVol > 0 Then
          sglWaterVolume = sglWaterVolume + sglInnunVol
     End If
     If sglWaterVolume > 136 Then
          sglWaterVolume = 136
     End If
     sglDensity = 998.34813 + 0.79884923 * sglPrevSalinity
_
          - 0.00436717 * sglPrevSalinity ^ (3 / 2) +
          0.00048314 * sglPrevSalinity ^ 2

'add/remove salt by innundation
     If sglInnunVol > 0 Then
          If sglTcSalinity > sglpwsalinity Then
               sglPwSaltMass = sglPwSaltMass +
               (sglTcSalinity * sglInnunVol * sglDensity /
               1000)
          Else: sglPwSaltMass = sglPwSaltMass -
          (sglTcSalinity * sglInnunVol * sglDensity / 1000)
          End If
          sglpwsalinity = sglPwSaltMass / (sglDensity /
          1000 * sglWaterVolume)
     End If

'remove salt by innundation/mixing

     If sglTcSalinity < sglpwsalinity And boolInnun = True
          And intSinceInnunDay > intDrainage And
          sglWaterVolume < 120 And sglR = 0 And
          sglPrecipitation = 0 Then
          sglPwSaltMass = ((sglTcSalinity + sglpwsalinity)
          / 2) * 136 * sglDensity / 1000
          sglpwsalinity = sglPwSaltMass / (sglDensity /
          1000 * sglWaterVolume)
     End If

'Add water from precip
     If intPrevInnunDay > intFieldCapacity Then
```

```
          sglWaterVolume = sglWaterVolume +
(sglPrecipitation * 2)
     Else
          sglWaterVolume = sglWaterVolume +
(sglPrecipitation * 0.12)
     End If

'Remove water by evaporation
     sglSalEvapCoefficient = 998.34813 / (998.34813 +
     0.79884923 * sglPrevSalinity _
           - 0.00436717 * sglPrevSalinity ^ (3 / 2) +
     0.00048314 * sglPrevSalinity ^ 2)
     sglEvaporation = sglEvaporation *
sglSalEvapCoefficient * 100

     If sglWaterVolume > invFieldCapacityVolume And
sglInnunVol < 0 Then
          sglWaterVolume = sglWaterVolume - sglEvaporation
     End If

     If sglWaterVolume > 136 Then
          sglWaterVolume = 136
     End If
'Remove water by drainage
      If sglInnunVol < 0 And n > 1 Then
          sglDrainWater = sglInnunVol * -1 * 0.00021 *
          ((sglWaterVolume / 136) / 0.68) ^ 23.8
           sglWaterVolume = sglWaterVolume - sglDrainWater

      'does not assume field capacity of 0.94 (morris 1995)
               If sglWaterVolume < 100 Then
                    sglWaterVolume = 100
               End If
            sglDensity = 998.34813 + 0.79884923 *
          sglPrevSalinity _
           - 0.00436717 * sglPrevSalinity ^ (3 / 2) +
          0.00048314 * sglPrevSalinity ^ 2
'Remove salt by drainage
        'assume field capacity of 0.89 model calibrated
               If sglWaterVolume > 122 Then
                    sglPwSaltMass = sglPwSaltMass -
                    (sglpwsalinity * sglDrainWater *
                    sglDensity / 1000)
               End If
```

```
            sglpwsalinity = sglPwSaltMass / (sglDensity /
          1000 *   sglWaterVolume)
        End If

'Salt crust mass balance

     sglDensity = 998.34813 + 0.79884923 * sglPrevSalinity
          _- 0.00436717 * sglPrevSalinity ^ (3 / 2) +
          0.00048314 * sglPrevSalinity ^ 2

     sglpwsalinity = sglPwSaltMass / (sglDensity / 1000 *
          sglWaterVolume)

'Remove salt forming a crust under prolonged emersion
     If intPrevInnunDay < 7 And boolInnun = False And
          sglWaterVolume > 112.2 And intPrevInnunDay >
          intFieldCapacity And sglWaterVolume < 136 And
          sglPrevSalinity > 10 Then
          sglDensity = 998.34813 + 0.79884923 *
          sglPrevSalinity _
                         - 0.00436717 * sglPrevSalinity ^
          (3 / 2) + 0.00048314 * sglPrevSalinity ^ 2
          sglRe = sglDensity * 0.025 * (sglWaterSurfaceArea
               ^ 0.5) / 0.00112
          sglK1 = 0.664 * sglRe ^ (1 / 2) * 500 ^ (1 / 3) *
               0.000173 / (sglWaterSurfaceArea ^ 0.5)
          sglC1Hat = 0.0137 * sglpwsalinity ^ 2 + 16.974 *
               sglpwsalinity

    'dblR should be nonnegative
          dblR = -1 * sglK1 * sglWaterSurfaceArea *
               (sglC1Hat - 5438.06) * 58.44277 / 1000
          sglCsBudgetMass = sglCsBudgetMass + dblR
          sglPwSaltMass = sglPwSaltMass - dblR
          sglDensity = 998.34813 + 0.79884923 *
          sglPrevSalinity _- 0.00436717 * sglPrevSalinity ^
          (3 / 2) + 0.00048314 * sglPrevSalinity ^ 2
          sglpwsalinity = sglPwSaltMass / (sglDensity /
               1000 * sglWaterVolume)


'add salt from crust following inundation
```

```
    Else If boolInnun = True And intSinceInnunDay <
         intFieldCapacity And intSinceInnunDay > 1 And
         sglInnunVol > 0 Then
         sglDensity = 998.34813 + 0.79884923 *
              sglPrevSalinity _
              - 0.00436717 * sglPrevSalinity ^ (3 / 2) +
              0.00048314 * sglPrevSalinity ^ 2
         sglRe = sglDensity * 0.025 * (sglWaterSurfaceArea
              ^ 0.5) / 0.00112
         sglK1 = 0.664 * sglRe ^ (1 / 2) * 500 ^ (1 / 3) *
              0.000173 / (sglWaterSurfaceArea ^ 0.5)
         sglC1Hat = 0.0137 * sglpwsalinity ^ 2 + 16.974 *
              sglpwsalinity
         dblR = sglK1 * sglWaterSurfaceArea * (sglC1Hat -
              5438.06) * 58.44277 / 1000

              If (-dblR) > sglCsBudgetMass Then
                        dblR = -sglCsBudgetMass
              End If

           dblCsBudgetMass = sglCsBudgetMass + dblR
           sglPwSaltMass = sglPwSaltMass - dblR
           sglDensity = 998.34813 + 0.79884923 *
              sglPrevSalinity _
              - 0.00436717 * sglPrevSalinity ^ (3 / 2) +
              0.00048314 * sglPrevSalinity ^ 2
         sglpwsalinity = sglPwSaltMass / (sglDensity /
              1000 * sglWaterVolume)
    ElseIf boolInnun = True And boolPrevInnun = True Then
         sglpwsalinity = sglpwsalinity
    End If
'Constrain negative outputs

    If sglPwSaltMass < 0 Then
         sglPwSaltMass = 0
    End If

    If sglWaterVolume < 111 And boolInnun = False And
         intPrevInnunDay > 8 And sglPrecipitation = 0 Then
         sglpwsalinity = 0
    End If

'Print data
    i = i + 1
```

```
        Worksheets("TimeSeries").Cells(i, 2) = sglpwsalinity
        Worksheets("TimeSeries").Cells(i, 3) = sglDensity
        Worksheets("TimeSeries").Cells(i, 4) = sglWaterVolume
        Worksheets("TimeSeries").Cells(i, 5) = sglPwSaltMass
        Worksheets("TimeSeries").Cells(i, 6) = sglDrainWater
        Worksheets("TimeSeries").Cells(i, 8) = sglInnunVol
        Worksheets("TimeSeries").Cells(i, 9) = boolInnun
        Worksheets("TimeSeries").Cells(i, 10) =intPrevInnunDay
        Worksheets("TimeSeries").Cells(i, 12) = dblR
        Worksheets("TimeSeries").Cells(i, 13) = sglK1
        Worksheets("TimeSeries").Cells(i, 14) = sglC1Hat
        Worksheets("TimeSeries").Cells(i, 15) = sglTcSalinity
        Worksheets("TimeSeries").Cells(i, 16) = sglEvaporation
        Worksheets("TimeSeries").Cells(i, 18) =sglCsBudgetMass
        Worksheets("TimeSeries").Cells(i, 19) = n
        Worksheets("TimeSeries").Cells(i, 20) =sglPrevSalinity
        Worksheets("TimeSeries").Cells(i, 21) =sglCsBudgetMass
        Worksheets("TimeSeries").Cells(i, 22)=sglPrecipitation

'Save data

        boolPrevInnun = boolInnun
        sglPrevSalinity = sglpwsalinity

'Reset variables

        sglDrainWater = 0
        dblR = 0

        If intSinceInnunDay = 3 Then
             sglCsBudgetMass = 0
        End If
Next n
```

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